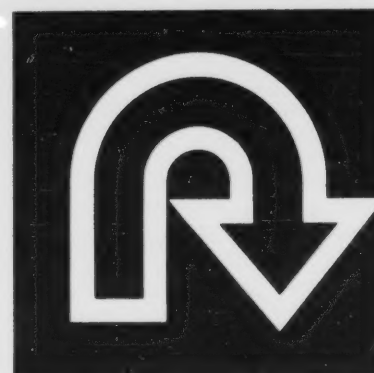


Interim
Target Loadings for
Acidic Deposition in Western Canada:
A Synthesis of Existing Information

WESTERN AND NORTHERN CANADA
LRTAP TECHNICAL COMMITTEE

March 1987



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Environmental Service

Canada

INTERIM
TARGET LOADINGS FOR
ACIDIC DEPOSITION IN WESTERN CANADA:
A SYNTHESIS OF EXISTING INFORMATION

FINAL REPORT

to

The Western Target Loading Planning Group
and
The National Research Council Associate Committee
on Scientific Criteria for Environmental Quality

by

ESSA Environmental and Social Systems Analysts Ltd.

and

Concord Scientific Corporation

Prepared with the supervision of:

The National Research Council
Associate Committee on
Scientific Criteria for
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for the

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Copies of this report are available from the Technical Committee:

Dr. R.G. Wilson (Chairman)	Ministry of Environment and Parks Waste Management Branch 810 Blanshard Street Victoria, British Columbia V8V 1X5 (604) 387-9987
Mr. R.F. Hopkinson (Secretary)	Environment Canada Atmospheric Environment Service Scientific Services, Regina P.O. Box 4800 Regina, Saskatchewan S4P 3Y4 (306) 780-5739
Mr. L.J. Lechner	Saskatchewan Environment and Public Safety Air Quality Branch Walter Scott Building 3085 Albert Street Regina, Saskatchewan S4S 0B1 (306) 787-6195
Dr. H.C. Martin	Environment Canada, LRTAP Liaison Office Atmospheric Environment Service 4905 Dufferin Street Downsview, Ontario M3H 5T4 (416) 667-4803
Dr. M. Morelli	Manitoba Environment and Workplace Safety and Health Environmental Standards and Studies Box 7, Building 2 139 Tuxedo Avenue Winnipeg, Manitoba R3N 0H6 (204) 945-7032
Dr. H.S. Sandhu	Alberta Environment Research Management Division 14th Floor, Standard Life Centre 10405 Jasper Avenue Edmonton, Alberta T5J 3N4 (403) 422-2071
Mr. M.M. Smith	Pollution Control Division Renewable Resources Government of the Northwest Territories Yellowknife, Northwest Territories X1A 2L9 (403) 873-7654

HISTORY

On October 25, 1984, western provinces and the federal government formally agreed to take cooperative action on the management strategy for -acidic deposition and decided to establish target loading(s) for sensitive ecosystems in western and northern Canada. In November 1984, J.P. Bruce [Assistant Deputy Minister, Environment Canada, and Chairman of the Western and Northern Canada Long-Range Transport of Airborne Pollutants (LRTAP) Consultative Committee] wrote to the President of the National Research Council (NRC) asking that NRC consider developing a scientific criteria document on atmospheric deposition for western Canada. The Management Subcommittee of Associate Committee on Scientific Criteria for Environmental Quality (ACSCEQ) proposed to carry out such a study on acid deposition in western Canada, and, in February 1985, the western provinces formally agreed to take cooperative action on a strategy to establish target loadings for western Canada.

In June 1985, Dr. D.R. Miller, NRC, prepared a proposal for the Western and Northern Canada LRTAP Consultative Committee identifying three components:

- 1) an initial document to be prepared by Canadian scientists knowledgeable in this area;
- 2) a workshop involving 10-12 experts (in addition to the Planning Group and representatives of LRTAP);
- 3) a final report specifically addressing the question of target loadings.

Overall supervision of this work was to be the responsibility of ACSCEQ, which struck a Planning Group:

Dr. H.H. Harvey (Chairman), University of Toronto
Dr. D.R. Miller (Project Manager), NRC
Dr. A.B. Cairnie (Project Manager in Dr. Miller's absence), Environment Canada
Dr. T.C. Hutchinson, Institute for Environmental Studies, University of Toronto
Dr. D.M. Whelpdale, Atmospheric Environment Service, Downsview
Dr. V.C. Runeckles, University of British Columbia
Dr. H.S. Sandhu, Alberta Environment

The western provinces and the Government of Canada contributed funding for specific activities:

British Columbia	\$10,000	Planning Meeting
Saskatchewan	\$10,000	Drafting Initial Document
Manitoba	\$10,000	Workshop
Alberta	\$10,000	Honoraria
Canada	\$12,250	Final Document

The Planning Group met at the Atmospheric Environment Service (AES) office in Downsview in December 1985, and wrote a draft contract and identified potential bidders. In January 1986, Requests for Proposals were sent to: Beak Consultants Ltd.; ESSA Ltd.; British Columbia Research Council; Alberta Research Council; Saskatchewan Research Council; Manitoba Research Council. The Planning Group was unanimous in supporting the ESSA/Concord proposal. On March 27, 1986, Contract #986-2101 was drawn up by NRC with ESSA/Concord to study and prepare a report on applicability of the Target Loading Concept to Western Canada.

The intent of the Planning Group, through Consultants, was to develop ranges of target loadings. After preliminary review of information it became clear that scientific information was limited. Considering the financial and time constraint, as well as lack of suitable scientific base, the Planning Group directed consultants to develop an approach to develop target loadings. This draft report fulfilled the terms of reference as agreed to between NRC's ACSCEQ Planning Group and Consultant.

A draft report was submitted by ESSA in August, 1986. This Report was reviewed by 32 experts, and a two-day workshop was held in Winnipeg, in August of 1986, to review comments received and to give clear direction to ESSA/Concord for completion of the final draft.

In March 1987, the NRCC Associate Committee on Scientific Criteria for Environmental Quality forwarded its report to the Technical Committee of Western and Northern Canada LRTAP.

Harold H. Harvey
for the Planning Group
March 1987

• • • •

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Final editing was done by Charles P. Newcombe, Senior Biologist with the Waste Management Branch, Ministry of Environment and Parks, Victoria, British Columbia. The final manuscript was produced by Fay Silcocks and Suzanne LeBlanc, both of the Waste Management Branch, Ministry of Environment and Parks, Victoria, British Columbia.

Dr. D.R. Miller
National Research Council of Canada
Associate Committee on Scientific
Criteria for Environmental Quality
Ottawa, Ontario

• • • •

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The Western Target Loadings Planning Group (consisting of Drs. Harold Harvey, Alan Cairnie, Larry Lechner, Hans Martin, Don Miller, and Clayton Rubec) was most supportive throughout the study, and gave many useful suggestions. The kind cooperation and flexibility of the scientific authorities, Dr. Cairnie and Dr. Miller, are much appreciated.

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ESSA Environmental and
Social Systems Analysts Ltd.
#705-808 Nelson Street
Vancouver, British Columbia
V6Z 2H2

and

Concord Scientific Corporation
2 Tippet Road
Toronto, Ontario
M3H 2V2

ACKNOWLEDGEMENTS (Continued)

Names and Affiliations of People who Contributed to the Draft Report

- | | |
|------------------------------|--|
| David Bezak, B.Sc. | - Manitoba Environment and Workplace Safety and Health, Winnipeg, Manitoba |
| Gordon Brown, M.Sc. | - University of British Columbia, Animal Resource Ecology, Vancouver, British Columbia |
| Alan B. Cairnie, Ph.D. | - National Research Council, Ottawa, Ontario |
| Lester W. Carlson, Ph.D. | - Canadian Forest Service, Government of Canada, Ottawa, Ontario |
| Malcolm J. Clark, Ph.D. | - Ministry of Environment and Parks, Victoria, British Columbia |
| Donald A. Faulkner | - Atmospheric Environment Service, Environment Canada, Vancouver, British Columbia |
| Betty Follows | - Ministry of Environment and Parks, Victoria, British Columbia |
| Glen H. Geen, Ph.D. | - Department of Biology, Simon Fraser University, Burnaby, British Columbia |
| Kathleen E. Grabowecky, B.A. | - Manitoba Environment and Workplace Safety, Winnipeg, Manitoba |

ACKNOWLEDGEMENTS (Continued) - Names and Affiliations

- | | |
|-----------------------------------|--|
| Harold H., Harvey, Ph.D. | - Department of Zoology,
University of Toronto,
Toronto, Ontario |
| Murthi S. Kotturi, Ph.D., P. Eng. | - Ministry of
Environment and Parks,
Victoria, British
Columbia |
| Larry J. Lechner, M.Sc., P. Eng. | - Saskatchewan
Environment,
Saskatoon,
Saskatchewan |
| Leonard E. Lillie, D.V.M., Ph.D. | - Alberta Environment,
Vegreville, Alberta |
| Samuel N. Linzon, Ph.D. | - Ontario Ministry of
the Environment,
Toronto, Ontario |
| Barid B. Manna, M. Eng., P. Eng. | - Ministry of
Environment and
Parks, Victoria,
British Columbia |
| K. Eric Marshall, B.Sc. | - Fresh Water Institute,
Department of
Fisheries and Oceans,
Winnipeg, Manitoba |
| Hans C. Martin, Ph.D. | - LRTAP Liaison Office,
Environment Canada,
Downsview, Ontario |
| Douglas G. Maynard, Ph.D. | - Northern Forestry
Centre, Canadian
Forest Service,
Government of Canada,
Edmonton, Alberta |
| David McCoy, M.Sc. | - Canterra Energy,
Calgary, Alberta |
| Susan J. Milburn, B.Sc. | - LRTAP Liaison Office,
Environment Canada,
Downsview, Ontario |

ACKNOWLEDGEMENTS (Continued) - Names and Affiliations

- | | |
|--------------------------------|---|
| Jeffrey L. Monty, B.Sc., RPF | - Petawawa National Forest Institute, Canadian Forest Service, Government of Canada, Chalk River, Ontario |
| David E. Moon, Ph.D. | - Land Resource Research Centre, Agriculture Canada, Vancouver, British Columbia |
| Max Morelli, Ph.D., P. Eng. | - Manitoba Environment and Workplace Safety and Health, Winnipeg, Manitoba |
| Charles P. Newcombe, M.Sc. | - Ministry of Environment and Parks, Victoria, British Columbia |
| David Onuczsko, B.Sc., P. Eng. | - Alberta Environment, St. Albert, Alberta |
| Craig J. Palmer, Ph.D., P. Ag. | - Alberta Environment, Lethbridge, Alberta |
| Dennis Parkinson, Ph.D. | - Kananaskis Centre for Environmental Research, University of Calgary, Calgary, Alberta |
| Gregory C. Pratt, Ph.D. | - Minnesota Pollution Control Agency, St. Paul, Minnesota |
| Michael G. Prior, Ph.D. | - Alberta Environment Centre, Alberta Environment, Vegreville, Alberta |
| Clayton D.A. Rubec, M.Sc. | - Inland Waters/Lands Directorate, Environment Canada, Hull, Quebec |

ACKNOWLEDGEMENTS (Continued) - Names and Affiliations

- | | |
|------------------------------|--|
| M. Hussain Sadar, Ph.D. | - Canadian Environment
Research Council,
Federal Environmental
Assessment Review
Office of Environment
Canada, Hull, Quebec |
| Harby S. Sandhu, Ph.D. | - Alberta Environment,
Edmonton, Alberta |
| John P. Senyk, B.Sc. | - Pacific Forestry
Centre, Canadian
Forest Service,
Government of Canada,
Victoria, British
Columbia |
| Peter W. Summers, Ph.D. | - Atmospheric
Environment Service,
Environment Canada,
Downsview, Ontario |
| Judith V. Thomas | - Research Management
Division, Alberta
Environment,
Edmonton, Alberta |
| Robert J. Vet, M.A.Sc. | - Atmospheric
Environment Service,
Environment Canada,
Downsview, Ontario |
| Douglas M. Whelpdale, Ph.D. | - Atmospheric
Environment Service,
Environment Canada,
Downsview, Ontario |
| John H. Wiens, Ph.D., P. Ag. | - Ministry of
Environment and Parks,
Victoria, British
Columbia |

ACKNOWLEDGEMENTS (Continued)

Reviewers of the Draft Report (including participants at the workshop held in Winnipeg, Manitoba, August 26-27, 1986 [Participants identified by *])

G. Barry Atkinson*, M.Sc.
Atmospheric Environment Service
Winnipeg, Manitoba

David J. Bezak, B.Sc. (Hon)
Manitoba Environment and
Workplace Safety and Health
Winnipeg, Manitoba

Thomas (Tom) G. Brydges*, Ph.D.
LRTAP Liaison Office
Environment Canada
Downsview, Ontario

William Dickson, Ph.D.
Statens Naturvardsverk
Stockholm, Sweden

Floyd C. Elder, M.Sc.
National Water Research Institute
Environment Canada
Burlington, Ontario

Pauline K. Erickson, B.Sc.
Conservation and Protection
Environment Canada
Edmonton, Alberta

Donald A. Faulkner*
Atmospheric Environment Service
Environment Canada
Vancouver, British Columbia

Eville Gorham, Ph.D.
Department of Ecology and
Behavioural Ecology
University of Minnesota
Minneapolis, Minnesota

Dennis J. Gregor, M.Sc.
Water Quality Branch
Environment Canada
Regina, Saskatchewan

ACKNOWLEDGEMENTS (Continued) - Reviewers and Workshop
Participants

Harold H. Harvey*, Ph.D.
Department of Zoology
University of Toronto
Toronto, Ontario

Arne Henriksen, Ph.D.
Norwegian Institute for
Water Research
Oslo, Norway

Nicholas Holowaychuk, Ph.D.
Terrain Sciences Department
Alberta Research Council
Edmonton, Alberta

Larry J. Lechner*, M.Sc.
Saskatchewan Environment
Regina, Saskatchewan

Marius A. Lusius, Ph.D.
Air Resources Branch
Ontario Ministry of the Environment
Toronto, Ontario

Bonnie L. Magill*, M.Sc.
Ministry of Environment
Edmonton, Alberta

Donald R. Miller*, Ph.D.
Environmental Secretariat
National Research Council
Ottawa, Ontario

Max Morelli*, Ph.D.
Manitoba Environment and
Workplace Safety and Health
Winnipeg, Manitoba

Charles P. Newcombe*, M.Sc.
Waste Management Branch
Ministry of Environment and Parks
Victoria, British Columbia

J. Stan Rowe*, Ph.D.
Department of Crop Science
and Plant Ecology
University of Saskatchewan
Saskatoon, Saskatchewan

**ACKNOWLEDGEMENTS (Continued) - Reviewers and Workshop
Participants**

Steven (Steve) C. Samis, B.Sc.
Department of Fisheries and Oceans
Vancouver, British Columbia

William W. Sawchin, Ph.D.
Fisheries Branch
Department of Parks, Recreation
and Culture
Saskatoon, Saskatchewan

David W. Schindler, Ph.D.
Department of Fisheries and Oceans
Fresh Water Institute
Winnipeg, Manitoba

John P. Senyk, B.Sc.
Pacific Forestry Centre
Canadian Forest Service
Government of Canada
Victoria, British Columbia

John W.B. Stewart, Ph.D.
Department of Soil Science
University of Saskatchewan
Saskatoon, Saskatchewan

Lester (Les) Swain, B.A.Sc.
Ministry of Environment and Parks
Victoria, British Columbia

Elon S. Verry, Ph.D.
USDA-Forest Service
Research Branch
North Central Forest Experiment Station
Grand Rapids, Minnesota, USA

David L. Wotton*, M.Sc.F.
Manitoba Environment and
Workplace Safety and Health
Winnipeg, Manitoba

John H. Wiens, Ph.D.
Waste Management Branch
Ministry of Environment and Parks
Victoria, British Columbia

Douglas M. Whelpdale*, Ph.D.
Atmospheric Environment Service
Downsview, Ontario

**ACKNOWLEDGEMENTS (Continued) - Reviewers and Workshop
Participants**

Paul H. Whitfield, M.Sc.
Inland Waters/Lands Directorate
Ministry of Environment
Vancouver, British Columbia

Richard (Rick) G. Wilson*, Ph.D.
Waste Management Branch
Ministry of Environment and Parks
Victoria, British Columbia

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1.0 INTRODUCTION

Increased recognition of adverse ecological consequences resulting from deposition of atmospheric contaminants onto continental areas has prompted the exploration of strategies to protect vulnerable ecosystems. Because existing environmental legislation is designed to regulate atmospheric emissions near the source, a commonly suggested strategy involves simply modifying this legislation to reduce atmospheric pollutant emissions. While seldom explicitly recognized, this strategy is based on dose-response concepts and assumes that to avoid undesirable ecological changes, the dose (atmospheric pollutant loadings) must be kept below some threshold value (a characteristic of the receptor ecosystem).

The maximum level of atmospheric deposition that affords long-term protection from adverse ecological consequences to all but the most sensitive portions of the ecosystem has been called the "target loading" (Memorandum of Intent 1983). A recent workshop on "Critical loads for effects of sulphur and nitrogen deposition on forest, soils, groundwater, and surface water" held in Norway during April, 1986, however, recommended using the term "critical loads" to refer to the highest deposition of pollutants that will not lead to harmful effects on biological systems. The authors of the "critical load" concept distinguish this idea from the earlier "target loading" by suggesting that the target load is what is both practically and politically achievable, while the critical load is the ultimate goal. Thus, although target loadings incorporate the notion of politically acceptable damage levels, critical loads are simply based on

ecosystem sensitivity. Even though the term target loading is used throughout this report, the reader should realize that in most cases what is referred to is, strictly speaking, more like the new "critical loading" concept.

Establishment of a target loading to protect ecosystems involves:

- 1) scientific research on atmospheric pollutants (sources, concentrations, interactions and deposition), and biotic receptor sensitivity (including related population frequency data);
- 2) risk analysis evaluating the dose-response relationship between pollutant concentrations and biotic effects (determination of the "critical loading"); and
- 3) examination of the socio-economic and political implications of any proposed target loading.

These three factors, conceptually modelled in Figure 1.1., fall into two broad categories: analysis and recommendation. The purpose of this report is to present the results of relevant scientific analyses, not to make specific recommendations on target loadings. Results from these analyses, however, should assist policy makers in formulating target loadings designed to protect ecosystems in western Canadian provinces and northern territories.

In this first section of the document, we summarize the history of concern about acid precipitation in western

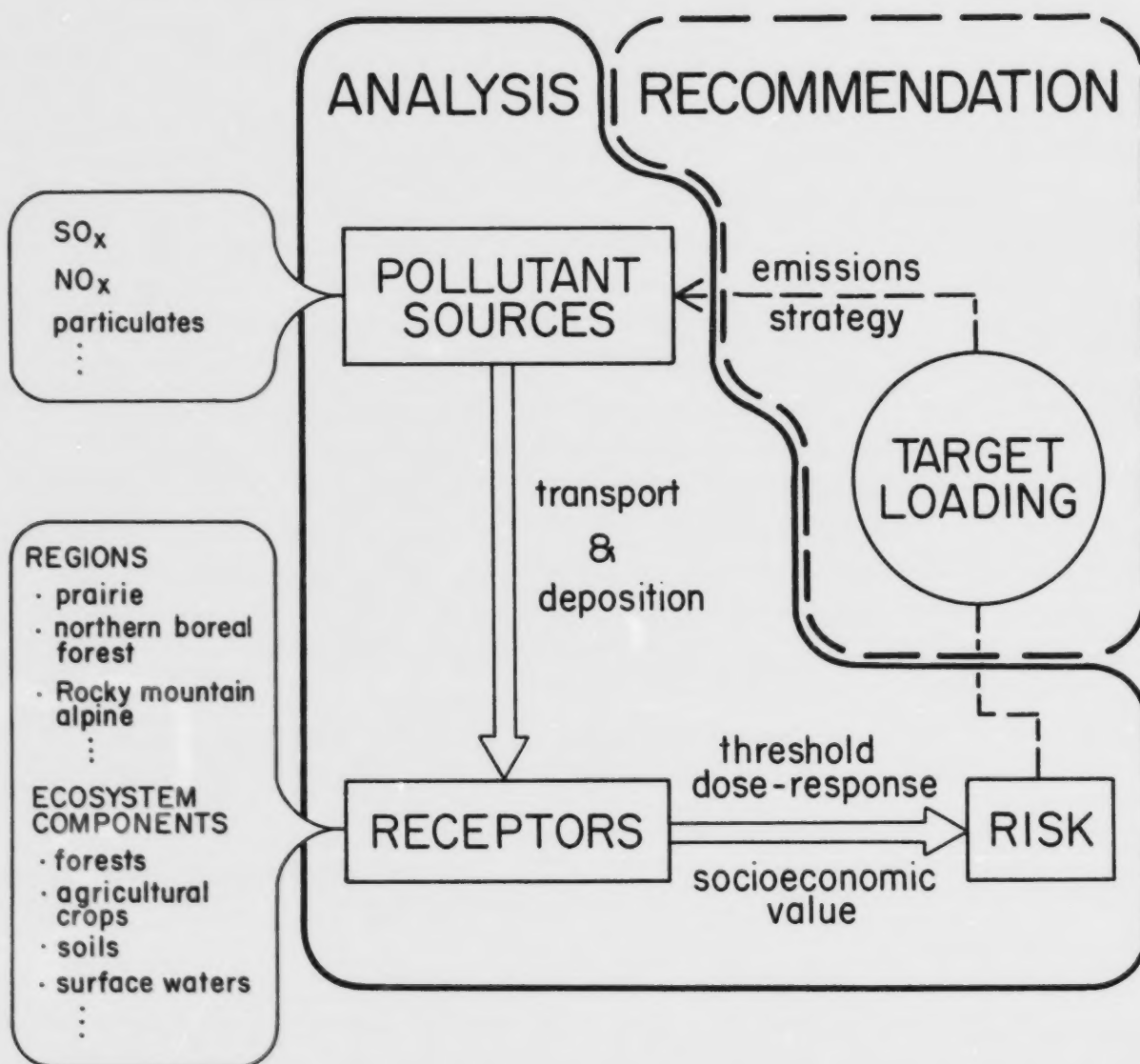


Figure 1.1: Simple conceptual model for establishing target loadings and concomitant emissions strategy. Note the importance of both dose-response relationships and socio-economic valuation in risk assessment.

Canada, review the methods that have been used to define target loadings, and describe the organization of the report's remaining sections.

1.1 Historical Overview

Until the early 1980s, both federal and provincial authorities did not consider acidic precipitation to be a major concern in western Canada. Then, in October, 1981, after conducting nation-wide hearings, the Subcommittee on Acid Rain of the Canadian Parliament released their report entitled "Still Waters: the Chilling Reality of Acid Rain". The authors of this report observed what they felt was a preoccupation with acidic precipitation issues in eastern Canada and suggested that additional funding and manpower be allocated to examining the situation in the western and northern portions of the country. Furthermore, the Subcommittee recommends the Government of Alberta accord maximum priority to the control of pollutants from industries in the province that cause acid rain. The Sub-committee recommends that the Provincial Government adopt as its guiding policy a goal of zero increase in emissions that cause acid rain over present levels up to the year 2000, and an annual decrease by a prescribed amount each year thereafter.

These recommendations and suggestions, along with others in the report, were largely based upon current knowledge of acidic precipitation in western Canada when the Subcommittee visited Calgary in February, 1981. At that time, submissions were heard from individuals representing sixteen different organizations; of the western provinces, only Saskatchewan made a formal presentation to the Subcommittee.

Although scientific data on precipitation chemistry were certainly available for many sites throughout this region, until this time only a few attempts (e.g., Summers and Whelpdale 1976; Hammer 1980) had been made to synthesize them and examine emergent regional patterns. One of the early attempts was made by Summers (1980:4) who concluded that there was "...no evidence for a widespread regional acid rain problem in the west either now or in the immediate future...". He did, however, go on to say that "... there are indications of more localized potential problems developing, especially where sensitive receptor areas lie downwind of the major sulphur dioxide sources". Summers recommended careful monitoring to assess the current situation and provide early warning of deteriorating conditions.

Meanwhile, government officials representing British Columbia, Alberta, Saskatchewan, Manitoba, the Northwest Territories, and Environment Canada formed two committees--a Consultative Committee and a Technical Committee to coordinate monitoring, research, and other activities related to acidic precipitation and long-range transport of air pollutants (LRTAP) in western Canada and northern Canada, (Sandhu and Wilson 1987). Under the aegis of these committees, a three-year research program was initiated in 1982 and six program priorities were established:

- 1) produce maps of aquatic sensitivity to acidic deposition;
- 2) produce maps of soil and geological sensitivity to acidic deposition;

- 3) monitor "background" precipitation chemistry;
- 4) model atmospheric deposition patterns;
- 5) inventory emissions of sulphur and nitrogen oxides;
and
- 6) monitor chemical changes in surface waters and soils
in data-sparse areas.

Studies carried out to date have been catalogued in the annual reports (1981-1984) released by the Western Canada LRTAP. While a brief synopsis of current findings associated with each project is given in these annual reports, until now no attempts have been made to synthesize these data.

In 1985, the Western LRTAP Consultative Committee contracted the Associate Committee on Scientific Criteria for Environmental Quality (ACSCEQ) of the National Research Council of Canada to investigate applicability of the target loading concept to Canadian regions west of the Manitoba-Ontario border. As part of this investigation, the ACSCEQ Planning Group contracted ESSA Environmental and Social Systems Analysts Ltd. (together with their associated consultants, Concord Scientific Corporation) to critically evaluate available data in light of establishing target loadings for western Canada.

ESSA and Concord together prepared a draft report, which was distributed by the National Research Council to be reviewed in detail by a distinguished group of scientists, including the Western LRTAP Consultative Committee, the

ACSCEQ Planning Group, and acidification experts from across Canada, the United States, Norway, and Sweden (listed in the Acknowledgements Section). In all, twenty-eight written reviews were received, with an average of twenty substantive comments per review. A workshop was held in Winnipeg, Manitoba on August 26-27, 1986 (attended by the participants listed in the Acknowledgements Section) to discuss the written reviews and recommend improvements to the document. Response to the draft report by reviewers was generally very positive. In this final report we have responded to all major concerns of the reviewers, and most of their minor concerns, to the maximum degree possible within the time and budget constraints of this study.

1.2 Alternative Methods for Defining Target Loadings

When establishing target loadings, the receptor ecosystem components of concern will greatly influence the analysis. For example, direct impingement of gaseous pollutants has the greatest effects on terrestrial ecosystem components (NAS 1981), whereas aquatic systems can be greatly affected by interactions between atmospheric pollutants and catchment geology. This means that while the chemical characteristics of underlying geological formations will have an important influence on aquatic sensitivity, they may be less important in determining effects (and thus target loadings) for terrestrial vegetation.

To date, target loadings have only been proposed for surface waters. No target loading criteria have been published for the protection of terrestrial systems, due to the large gaps in scientific understanding of dose-response relationships. Target loadings for surface waters have been

based upon empirical models and observations of coincident effects, suggesting acidification processes are initiated when atmospheric loadings of sulphate or hydrogen ions exceed certain levels. After reviewing evidence for dose-response acidification relationships, Evans et al. (1981) concluded that an annual volume-weighted hydrogen concentration of 25 eq·L⁻¹ (pH = 4.6) in precipitation appeared to be a critical threshold. This proposed target loading was rejected, however, by the authors of the Memorandum of Intent (MOI 1983; page 3-213) who argued that target loadings should be based on sulphur deposition rather than hydrogen ion loading.

This suggestion was founded upon empirical observations indicating that H⁺ concentrations and loadings bear little relationship to the acidifying potential of precipitation. The use of H⁺ as the preferred ion for defining target loadings was rejected because of the biosystem's differential capacity to assimilate nitrate, ammonia, or sulphate ions (the ions in precipitation primarily responsible for acidification) and because of the biosystem's differential responses to these ions (MOI 1983; page 3-213).

Since sulphate is the predominant ion associated with acidification, the authors of the Memorandum of Intent (MOI) used both the empirical approach of Henriksen and the theoretical model of Thompson (1982) to propose a target loading of 20 kg of sulphate per hectare per year. They considered this loading limit sufficient to keep most surface waters above pH 5.3, excepting more sensitive waters in areas of low runoff (MOI 1983; pages 3-204 to 3-215).

Other investigators have suggested limits lower than 20 kg per hectare per year. Gorham et al. (1984) analyzed the

composition of wet deposition across the eastern half of the United States and observed that a loading of 20 kg of wet sulphate $\text{SO}_4^{=}$ per hectare per year would produce an average pH in precipitation of 4.44, well below the threshold pH of 4.6 recommended by Evans et al. The statistical analyses of Gorham et al. suggested that, for the eastern U.S., annual loadings of sulphate in the range of 14 - 16 kg per hectare would maintain precipitation pH in the range of 4.7 to 4.6 (respectively). (These recommended loadings were computed assuming an annual precipitation of 88.7 cm; at 1 m precipitation, the loading levels would be 16 - 18 kg per hectare.) In a recent article, Newcombe (1985) describes an empirical approach to setting target loadings which correlates deposition with biological impacts, and suggests a limit of between 10 and 20 kg of sulphate per hectare per year.

A recent study by the Minnesota Pollution Control Agency (MPCA) (1985) is very pertinent to the issue of determining target loadings. The MPCA used a number of different methods to estimate target loadings for aquatic systems, and then compared the results. The methods included simulation models, empirical models, and empirical comparisons across regions receiving different levels of deposition. Table 1.1 summarizes the results of the Minnesota study. The models applied in Minnesota are transferable only to regions with similar climate, geology, soils, and deposition. After determining the minimally acceptable pH for precipitation was 4.7, the MPCA used a nonlinear regression equation to relate sulphate and hydrogen ion concentrations in Minnesota precipitation. This method led them to conclude that a deposition limit of 11 kg of sulphate $\cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ should be

TABLE 1.1

Summary table of recommended precipitation pH levels, or sulphate target loadings, or both, to prevent lake acidification. (Source: Minnesota Pollution Control Agency 1985.)

SOURCE	SULPHATE TARGET LOADING OR pH STANDARD	COMMENTS
Almer et al. 1978	15-17 kg·ha ⁻¹ ·yr ⁻¹ (total)	Lakes with alkalinities between 100-200 µeq·L ⁻¹ would lose alkalinity and decline in pH. Very sensitive lakes would be expected to acidify.
	9-12 kg·ha ⁻¹ ·yr ⁻¹ (total)	Very sensitive lakes would be protected from acidifying; no degradation (loss of alkalinity or pH) in lakes with alkalinities between 100-200 µeq·L ⁻¹ .
Henriksen 1980	20 kg·ha ⁻¹ ·yr ⁻¹ (wet)	Majority of sensitive lakes would not acidify; this loading is associated with a precipitation pH of 4.5. Highly sensitive lakes would most likely acidify.
	10-12 kg·ha ⁻¹ ·yr ⁻¹ (wet)	Highly sensitive lakes with alkalinities less than 50 µeq·L ⁻¹ would be protected from acidification.
Brydges and Neary 1984	20 kg·ha ⁻¹ ·yr ⁻¹ (wet)	The majority of sensitive lakes will not acidify. Lakes with alkalinities less than 50 µeq·L ⁻¹ may acidify.
	< 15 kg·ha ⁻¹ ·yr ⁻¹ (wet)	Protect very sensitive lakes from acidifying. The majority of sensitive lakes would not show any significant effects from acid deposition.
Wright 1983	pH 4.7	Protects lakes with (Ca + Mg) concentrations greater than 40 µeq·L ⁻¹ from acidifying.
Oppenheimer 1984	18 kg·ha ⁻¹ ·yr ⁻¹ (wet)	The majority of sensitive lakes will be protected from acidifying. Very sensitive lakes may acidify.
	< 18 kg·ha ⁻¹ ·yr ⁻¹ (wet)	Very sensitive lakes will be protected from acidifying. The majority of sensitive lakes will probably not show any major effects from acid deposition.
Rogalla and Brezonik 1985	pH 4.58-4.78	Protects lakes with alkalinities greater than 45 µeq·L ⁻¹ from acidifying.
Schnoor 1984	pH 4.6	Protects the most sensitive lakes (seepage lakes) from acidifying.
MPCA 1985	pH 4.3-4.4 (14-15 kg·ha ⁻¹ ·yr ⁻¹ wet)	Protects sensitive lakes with alkalinities greater than 60 µeq·L ⁻¹ from acidifying. Lakes with alkalinities less than 60 µeq·L ⁻¹ will most likely acidify.
	pH 4.7 (11 kg·ha ⁻¹ ·yr ⁻¹ wet)	Protects the very sensitive lakes, alkalinities between 40-60 µeq·L ⁻¹ , from acidifying. Prevents loss of alkalinity and pH in lakes with alkalinities between 60-100 µeq·L ⁻¹ .

applied within a designated region containing sensitive aquatic resources. An averaging period of less than one year was considered, but dismissed as being administratively unworkable. The important relationship between sulphate and hydrogen ion concentrations in precipitation varies from region to region (Church and Galloway 1984), and therefore, data specific to the region must be used to determine what sulphate loading level corresponds to the assumed threshold concentration. Sections 2 and 3 of this report pay special attention to this issue.

The issue of whether to use concentration limits or total loading limits to protect ecosystems is still unresolved. At a recent workshop on critical loadings (held in Norway in April, 1986), Henriksen et al. (in prep.) suggested that a concentration limit (either pH or sulphate) may be preferable to a deposition limit, because: a) aquatic effects appear to be more strongly related to concentration than to loading; and b) a concentration limit has wider application than a deposition limit, since precipitation levels (and therefore total loadings) are so variable regionally. Henriksen's group proposed four different methods for estimating critical loadings:

- 1) graph lake pH versus wet deposition of sulphur using different curves for sets of lakes with different bedrock and soils (classified by the combined concentrations of calcium and magnesium in lakewater);
- 2) calculate the level of acidic anion inputs which would not exceed the base cation supply (computed

from runoff, lake concentrations of base cations, and an assumed level of cation exchange);

- 3) set a limit on the concentration of sulphate in runoff, equal to the background concentration of sulphate in sensitive lakes (i.e., those not exposed to acidic deposition); and
- 4) graph the pH ranges observed in a set of streams against stream concentrations of non-marine sulphate, noting with each data point (median pH), the median value for the sum of non-marine calcium plus magnesium.

The potential for application of these methods, and others, to western Canada is examined in considerable detail within Sections 4, 5, 6, and 7 of this report. Three of the major issues of concern are the feasibility of using wet deposition of sulphate as a measure of acid loading, the lack of appropriate methods and data for defining target loadings to terrestrial systems, and the uncertainty in the aquatic resource at risk within each region.

Ideally, one would develop target loading standards by summing up the damages associated with a given level of deposition across all receptor components (i.e., soils, crops, forests, surface waters, wetlands), or select the most sensitive component within each region based on dose-response relationships. In the Minnesota study, surface waters were considered to be more sensitive than other receptors and the standard for protection of aquatic systems was thought to be sufficient for protection of soils, forests, and wetlands.

1.3 Report Organization

The remaining sections of this report roughly correspond to the major components of the conceptual model shown in Figure 1.1. Chapters 2 and 3 describe sources and concentrations of important air pollutants in western Canada as well as the adequacy of existing databases for estimating pollutant loadings. Chapter 4 provides a regional description of ecological receptors and their sensitivity to atmospheric pollutants. A detailed summary of current knowledge pertaining to the responses of ecosystems to known doses of air pollutants is provided in Chapter 5. Chapter 6 examines the potential for applying the target loading concept to western Canada. Finally, major uncertainties and recommendations are summarized in Chapter 7. References Cited, and a Glossary of Acronyms, Models, Symbols, and Specialized Terms are in Chapters 8.0 and 9.0 respectively.

2.0 AIR POLLUTANTS OF IMPORTANCE IN WESTERN CANADA

The range of airborne pollutants in western Canada is typical of those found elsewhere, and includes gaseous sulphur and nitrogen oxides, hydrocarbons, photochemical oxidants, reduced sulphur compounds, suspended particulate matter, together with other organic and inorganic compounds, such as trace elements.

Traditionally, the pollutants that have been considered most important for the establishment of target loadings are the sulphates and nitrates in wet deposition. These compounds are, of course, derived from the primary sulphur and nitrogen oxide pollutants, and are believed to be indicative of the acidic nature of deposition. Also, the wet component of the deposition has been used as a surrogate for total deposition. Although these pollutants are likely to be most important in western Canada also, we will examine the rationale by which the importance of pollutants in western Canada may be judged. Firstly, we will establish criteria to define importance and then apply the criteria in the context of western Canada.

2.1 Criteria to Define Important Air Pollutants

One of the fundamental implications of the establishment of a target loading is the existence of a stress (in this case, a dosage of airborne pollutants) on one or more receptors. Criteria defining 'importance' may be based on a variety of subjective considerations. Without rigorous treatment, we will presume that the criteria should address the following aspects:

- 1) the amounts of emissions in western Canada relative to total Canadian emissions;
- 2) the relative contributions from natural and manmade sources in western Canada; and
- 3) the potential for adverse effects.

The emissions in western Canada are viewed in the context of the total Canadian emissions in order to provide a frame of reference. The second aspect implies not only that emissions from manmade sources should be 'significant' in comparison with natural sources, but also that the resulting ambient levels and hence the dose or loading pose an existing or foreseeable threat to receptors deemed worthy of protection. The third aspect, (3), assumes knowledge of the nature and extent of receptors at risk and the likelihood that adverse effects may result from exposure to pollutants in (1).

These considerations suggest the following criteria:

- 1) Emissions levels from manmade sources should be significant relative to those from natural sources.
- 2) Levels of the pollutants in ambient air and in precipitation should be known and should have potential for harmful effects.
- 3) Receptors likely to be affected by the pollutants should exist in the study area.

- 4) The potential for increased ambient levels should exist and that such levels may approach doses that may cause adverse effects.

Each of the pollutants listed earlier will be examined in terms of the criteria in order to establish their importance in the western Canadian context.

In applying the criteria, we will consider the following classes of compounds:

- sulphur compounds (includes sulphur oxides, hydrogen sulphide and other reduced sulphur compounds, particulate sulphates);
- nitrogen compounds (includes nitrogen oxides, ammonia, particulate nitrates and ammonium compounds);
- suspended particulate matter (includes compounds derived from soil, especially calcium and magnesium);
- photochemical oxidants (ozone and related organic compounds); and
- organic compounds (includes volatile hydrocarbons and toxic organic compounds such as polynuclear aromatic hydrocarbons, and pesticides).

2.2 Assessment of Important Compounds

2.2.1 Sulphur Compounds

Emissions

The manmade and natural emissions of sulphur dioxide in western Canada (and totals for Canada) for the year 1980 and projections for the year 2000 (MOI, 1982a) are summarized in Table 2.1. The MOI emissions data represent the most consistent data insofar as the data were derived using the same methodology and refer to the same year. Although the Memorandum of Intent (MOI, 1982a) forecasts increases in Alberta's sulphur dioxide emissions, other observers (Colley and Poone 1982) forecast decreases by the year 2000.

A Canadian emissions inventory for 1980 has been published recently (EPS, 1986) but was unavailable during preparation of this report. More recent emissions data are available for some provinces, but different methodologies were used for compilation and data are for different years. These data are given in Table 2.2. Also indicated in Table 2.2 are estimates of manmade emissions for the year 2000 for Alberta, Saskatchewan and Manitoba. It should be noted that Manitoba is party to a 1985 agreement among eastern provinces and the federal government to reduce SO₂ emissions by 1994 to 50% of the 1980 emissions. Under this agreement, the allowable (regulated) SO₂ emissions in Manitoba would decrease from 738,000t in 1980 to 550,000t in 1994. Information on major point sources in each province is given in Appendix I.

It should be noted that the estimated emissions in the year 2000, given in Table 2.1, were made in 1979-1980. These

Table 2.1

Emissions of SO₂ (10⁶ tonnes) in western Canada, and Canada, for 1980 and 2000¹

Region	Year 1980			Year 2000		
	Manmade	Natural	Total	Manmade	Natural	Total
British Columbia	0.191	0.083	0.274	0.249	0.083	0.332
Alberta ²	0.539	0.066	0.605	0.594	0.066	0.660
Saskatchewan	0.058	0.042	0.100	0.049	0.042	0.091
Manitoba	0.490	0.103	0.593	0.754	0.103	0.847
N.W.T./Yukon	0.002	0.028	0.030	0.002	0.028	0.030
Western Canada ³	1.278	0.294	1.572	1.646	0.294	1.940
Canada	4.8	0.639	5.439	4.51	0.639	5.149

¹ Data from U.S.-Canada Memorandum of Intent (MOI, 1982a).

² Although the Memorandum of Intent (MOI, 1982a) forecasts increases in Alberta's sulphur dioxide emissions, other observers (Colley and Poone 1982) forecast decreases by the year 2000.

³ Excludes Yukon and N.W.T.

Table 2.2
Provincial SO₂ emissions

Region	Emissions (tonnes)	Year	References
British Columbia	205,000	1982	B.C. MOE 1985
Alberta	559,600	1981	Alberta Environment 1982
	743,500 ^a	2000	
	535,500 ^b	2000	
	225,900 ^c	2000	
Saskatchewan	97,500	1982	Howard and Underwood 1984
	100,000	2000	
Manitoba	423,000	1981	Campbell 1984
	738,000 ^d	1980	
	500,000 ^d	1994	

- a. Present technology, low growth scenario (Colley and Poon 1982).
- b. Best practicable technology, low growth scenario (Colley and Poon 1982).
- c. Best available technology, low growth scenario (Colley and Poon 1982).
- d. Allowable or regulated emissions.

estimates were based largely on projections of population and the level of economic activity, of which the latter is fraught with considerable uncertainty.

Emissions data for western Canada may be compared with those for the remainder of Canada by examining the absolute amount and the emissions density (emissions per unit area) for the year 1980 (See Appendix II for data on areas).

The SO_2 emissions in western Canada are about 25% of the Canadian total. The average SO_2 emissions density (expressed as $\text{tonnes} \cdot \text{km}^{-2} \cdot \text{y}^{-1}$) for the Provinces in western Canada (British Columbia, Alberta, Saskatchewan, Manitoba) is $0.43 \text{ tonnes } \text{SO}_2 \cdot \text{km}^{-2} \cdot \text{y}^{-1}$ while for the eastern Canadian provinces, the emissions density is $1.13 \text{ tonnes } \text{SO}_2 \cdot \text{km}^{-2} \cdot \text{y}^{-1}$. The emissions density for the four western Canada provinces is thus, on average, at least 2.6 times lower than for eastern Canada. Figure 2.1 shows ranges of the emissions density for SO_2 emissions in 1980, on a 127 km CMC grid. It should be noted that the Yukon and Northwest Territories were excluded from the calculation of the emissions density because of their large area and very low emissions. The effect of the high emissions density in eastern Canada is exacerbated by relatively large influxes from the U.S., a situation which does not exist in western Canada where only in British Columbia has it been shown that some transport from the US affects that province. Based on projections, the SO_2 emissions density in western Canada will increase by ~26% by the year 2000. At that time, the emissions density would still not approach current (1980) levels in eastern Canada [U.S.-Canada Memorandum of Intent (MOI) 1982].

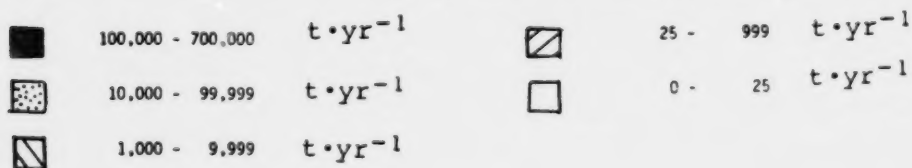
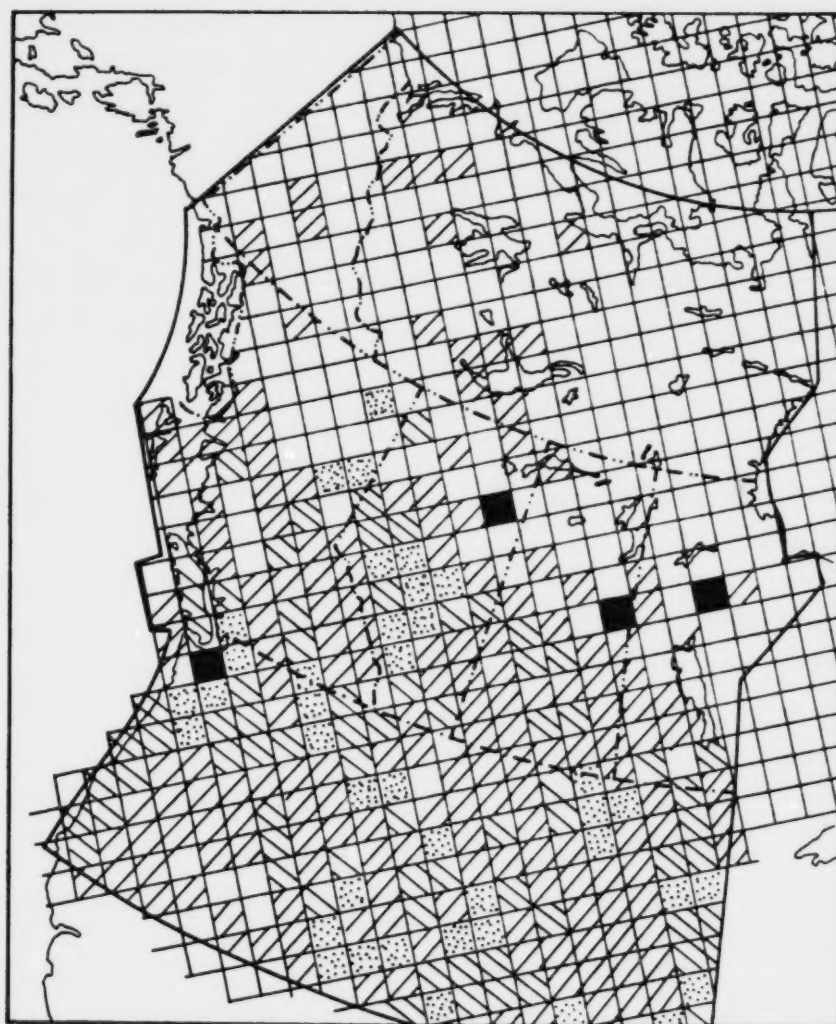


Figure 2.1: SO₂ emissions inventory (1980) on a 127 km grid for western Canada and contiguous states in the U.S.

Information on the amounts of natural sources of sulphur oxides is not very accurate, but manmade emissions (~80% of the total) are much more important than natural emissions. The major natural sources of SO_x are volcanic action, and other natural phenomena associated with geological activity (geysers and hot springs for example). It should be noted that emissions inventories listed include SO_2 emissions from forest fires in the manmade sources, but this source is typically small (forest fire sources of SO_2 are of the order of 0.04% of the manmade sources).

Emissions of hydrogen sulphide are summarized in Table 2.3. In contrast to SO_2 , the natural emissions are much larger than the manmade emissions. The total H_2S emissions in western Canada are small compared to SO_2 (0.18×10^6 tonnes $\cdot\text{y}^{-1}$ for H_2S versus 1.57×10^6 tonnes $\cdot\text{y}^{-1}$ for SO_2) but are by no means negligible. However, it should be noted that there is considerable uncertainty in the emissions data for H_2S . For example, estimates of natural H_2S emissions could be 30-50% too high (CSC 1985).

The atmospheric chemistry of H_2S is such that it will undergo conversion to oxidized sulphur species relatively quickly. Estimates of the residence times for H_2S based on various atmospheric situations range from 2 h to 3.5 days (CSC 1985). Appendix III lists these residence times. It is likely that H_2S will eventually contribute to SO_2 and sulphate deposition as a result of the atmospheric reactions of H_2S .

Hydrogen sulphide emissions are relatively more important in western Canada Provinces than in eastern Canada

Table 2.3

Emissions of hydrogen sulphide in Canada
(tonnes·y⁻¹)¹

Region	Natural	Manmade	Total
British Columbia	40,790	8731	49,521
Alberta	31,717	264	31,981
Saskatchewan	19,652	762	20,414
Manitoba	50,510	664	51,174
N.W.T.	22,862	N	22,862
Yukon	4,950	N	4,950
Western Canada ²	170,481	10,421	180,902
Western Canada ³	142,669	10,421	153,096
Eastern Canada	139,614	12,284	151,898
Canada	310,095	22,705	332,800

N Negligible.

¹ CSC, 1986. (Based on: National Research Council of Canada. 1981. Hydrogen sulfide in the atmospheric environment: scientific criteria for assessing its effects on environmental quality. National Research Council of Canada. NRCC No. 18467.)

² Includes Yukon and Northwest Territories.

³ Excludes Yukon and Northwest Territories.

important in western Canada Provinces than in eastern Canada since:

- 1) the H_2S emissions density is higher ($0.062 \cdot \text{t km}^{-2} \cdot \text{y}^{-1}$ in western Canada versus $0.048 \cdot \text{t km}^{-2} \cdot \text{y}^{-1}$ in eastern Canada); and
- 2) the ratio of H_2S to SO_2 in western Canada is greater than in eastern Canada (0.23 and 0.078 respectively, expressing H_2S as SO_2).

Since the total H_2S emissions in western Canada could be 23% of the total SO_2 emissions in western Canada (expressing H_2S as SO_2 , assuming complete conversion to SO_2), more information on the atmospheric chemistry of H_2S should be obtained. In addition, there is a requirement for research efforts directed at quantifying emissions of H_2S and other reduced sulphur compounds and assessing the extent to which they contribute to acidic deposition.

Effects

The effects of acid-forming sulphur compounds on aquatic and terrestrial receptors will be discussed in subsequent sections of the report here. It is clear that since acid-forming emissions are present, the potential for damage should be examined. The likely effects are due to direct impacts on vegetation (injury, loss in yield) or more indirect ecosystem effects (e.g., lake acidification, and leaching of plant nutrients from the soil).

Ambient Levels

Knowledge of the ambient levels of sulphur compounds is available mainly from monitoring stations located near industrial plants or from urban networks. In the case of plant-specific air quality monitoring, the incidence of violations of ambient air quality standards would suggest a potential for short-term damage near industrial plants. Data on ambient air quality levels in rural (i.e., agricultural or forested) areas in western Canada are sparse.

Published summaries of province-wide occurrences of concentrations in excess of air quality standards for sulphur compounds (SO_2 and H_2S) near industrial sources are generally not available, although information is publicly accessible. Published summaries of air quality data from urban stations (federal and provincial networks) are available but have limited use in the context of target loading, since stations are in urban areas and are not representative of the larger rural areas.

In the case of precipitation data, the provincial and federal precipitation networks provide wet deposition measurements throughout the region. Most stations are in non-urban locations but not all stations would satisfy the criteria for being regionally representative. Some stations provide information on precipitation chemistry in the region surrounding large point sources (e.g., stations in northern Saskatchewan and in Manitoba).

Published summaries of data from the precipitation networks are available but apart from the published data for

1980, more recent analyses of the data for the region as a whole are lacking.

2.2.2 Nitrogen Compounds

Emissions

Recent (1980) and projected (2000) emissions of nitrogen oxides (expressed as NO_2) for western Canadian provinces and the total for Canada, are listed in Table 2.4. More recent provincial estimates are given in Table 2.5.

Data in Table 2.4 show that the western Canada NO_x emissions are ~48% of the total Canadian emissions. Expressed in terms of emissions density, western Canada has $0.27 \text{ tonnes } \text{NO}_x \cdot \text{km}^{-2} \cdot \text{y}^{-1}$. The NO_x emissions in the year 2000 are projected to increase by ~34% above 1980 levels. Similar increases are projected for western and eastern Canada; therefore, there is the potential for increasingly significant effects by nitrogenous air pollutants.

In western Canadian provinces, the manmade NO_x emissions are a significant fraction (0.48) of the total NO_x emissions.

In contrast to the case of SO_2 , the largest portion of NO_x emissions arises from transportation, which corresponds to a low elevation area source located in urban centres. In general, the only significant point source emitters of NO_x are thermal power plants. Only in the case of Alberta are there significant NO_x sources other than thermal power

Table 2.4

Emissions of NO_x (as 10⁶ tonnes NO₂) in western Canada
and Canada for 1980 and 2000¹

Region	Year 1980			Year 2000		
	Manmade	Natural	Total	Manmade ²	Natural	Total
British Columbia	0.173	0.232	0.504	0.272	0.232	0.504
Alberta	0.354	0.130	0.484	0.629	0.130	0.759
Saskatchewan	0.148	0.214	0.362	0.239	0.214	0.453
Manitoba	0.078	0.242	0.320	0.109	0.242	0.351
N.W.T./Yukon	0.026	0.350	0.376	0.038	0.350	0.388
Western Canada ³	0.753	0.818	1.561	1.749	0.818	2.067
Canada	1.800	2.690	4.490	2.420	2.690	5.110

¹ MOI, 1982c.

² Projections for the year 2000 are based on the assumption that vehicle emissions standards for the reduction of NO_x are not in place. Projections that assume that vehicle emissions standards are in place would be lower than those shown here.

³ Excludes Yukon and Northwest Territories (N.W.T.).

Table 2.5

Summary of provincial NO_x emissions

Region	Emissions (tonnes)			Transportation Contribution (%)
	Province	(year)	MOI ⁶	
British Columbia	165,000 ¹	(1978)	172,600	72
Alberta	339,800 ² 363,340 ³	(1980) (1982)	353,500	51
Saskatchewan	127,400 ⁴	(1982)	148,300	65
Manitoba	73,800 ⁵	(1981)	78,200	84
Yukon/N.W.T.			26,300	72

- References:
- ¹ BC MOE 1985.
 - ² Alberta Environment 1982.
 - ³ Nitrogen oxide emissions for Alberta, 1978-1982. Alberta Environment, Air Quality Control Branch, March 1984.
 - ⁴ Howard and Underwood 1984.
 - ⁵ Environmental Management Division, undated report.
 - ⁶ Data for 1980. MOI, 1982a.

plants or transportation sources. The extensive network of compressor stations required for the movement of natural gas in pipelines results in numerous small point sources of NO_x that are located in rural areas.

Information on emissions of other nitrogen compounds (e.g., NH_3) is sparse since emissions inventories are not available. In the case of NH_3 emissions (e.g., from farming activity) a neutralizing effect through reaction with acidic species and resulting in the formation of ammonium compounds, would occur. The nitrification (oxidation) of the ammonium compounds, however, results in acidification.

Effects

The better known and probably more important effects of nitrogen oxides on various receptors are those caused by acidification in wet or dry deposition rather than by direct injury through exposure to gaseous NO_x .

Ambient Levels

Ambient levels of NO_2 and NO_x (total nitrogen oxides) are monitored in the vicinities of industrial plants (compressor stations) and in urban networks. These data have limited use in the context of target loading since the stations are not in regionally representative locations. Precipitation data for nitrates parallel the data for sulphur compounds: the data for nitrate in precipitation came from provincial and federal networks, as well as in stations northern Saskatchewan and in the vicinity of large point sources in northern Manitoba.

2.2.3 Photochemical Oxidants

Emissions

Photochemical oxidants are secondary pollutants resulting from atmospheric reactions involving nitrogen oxides, sunlight, and hydrocarbons. In view of the complex nature of the relationships between ozone and its precursors (NO_x , reactive hydrocarbons), emissions inventory information for hydrocarbons will not be summarized here since there is no simple relationship between ozone levels and hydrocarbon emissions.

Effects

In the context of this study, ozone effects on vegetation (as opposed to human health) are relevant. Extensive dose response data for ozone on crops are available and generally indicate that there is potential for yield loss at growing season mean ozone levels in excess of 'background' levels of ~ 0.025 ppm. The reported ozone levels in rural Alberta (Angle and Sandhu 1986) and in the Lower Fraser Valley in British Columbia (CSC 1982) clearly suggest that ozone levels may exist at concentrations likely to elicit measurable crop losses.

Based on these data, further investigation into the regional distribution of ozone levels in western Canada is warranted. In terms of target loading, the loading needs to be expressed entirely in terms of dry deposition and would depend heavily on reliable and area-specific dose-response functions.

Ambient Levels

Data for ambient ozone levels that are regionally representative of areas in western Canada are very sparse. Existing federal and provincial ozone monitoring stations are primarily in urban centers. Limited data on rural ozone measurements in Alberta however, showed that one-hour ozone levels can approach and occasionally exceed provincial (Alberta, British Columbia) standards. [Data from Birch Mountain, Bitumont and Ellerslie in Alberta (Angle and Sandhu 1986) or from Seymour Dam, Abbotsford and Chilliwack in British Columbia (CSC 1982)].

2.2.4 Particulate Matter

Emissions

Particulate emissions and the resulting air quality measurements of total suspended particles are important in terms of adverse effects such as poor visibility and health effects. In the present context, however, particles have the potential for complicating wet and dry deposition measurements of acid forming pollutants. Thus, alkaline particulate matter such as calcium, magnesium, and ammonium salts as well as particulate sulphate and nitrates derived from natural sources--wind blown dust or sea spray--or from manmade sources, can dramatically affect the apparent acidity of wet deposition. Information on alkaline natural particulate and sulphate emissions are, therefore, discussed below.

Emissions inventory data for natural sources of the alkaline particulate matter containing the metals, Ca, Mg,

and K in western Canada are summarized in Table 2.6. Sea spray emissions of sulphate are given in Table 2.7.

The emissions of alkaline particulate matter need to be viewed in relation to the SO_2 and NO_x emissions as well as in relation to ambient measurements of suspended particulate matter. Such comparisons provide very crude indications of whether or not neutralization of acidic sulphur and nitrogen species could possibly be affected by alkaline particulate matter.

The sum of the calcium and magnesium emissions from natural sources is 7.0×10^{10} equivalents $\cdot \text{y}^{-1}$ and may be compared with total manmade emissions of SO_2 and NO_x of 8.2×10^{10} equivalents y^{-1} . Of the total for Ca and Mg, about 5.5% (Environmental Applications Group 1982) would be in the size fraction less than $10\mu\text{m}$ (i.e., capable of remaining airborne and, therefore, transportable over long distances and susceptible to being washed out in wet deposition). Mechanisms which allow the introduction of larger particles in deposition samples, thereby resulting in interference with the acidity of wet deposition in western Canada, are possible but have not been demonstrated. The amount of small ($<10\mu\text{m}$) alkaline particles is, therefore, of the order of 4.5% of the total amount of the acid equivalents of SO_2 and NO_x emissions. Because of the logarithmic nature of the pH scale, it is conceivable that alkaline particulate salts of calcium and magnesium could have a significant impact by elevating the pH levels in wet deposition in western Canada.

The importance of Ca^{++} , Mg^{++} , and NH_4^+ in the precipitation chemistry at western Canada CANSAP sites was recently

Table 2.6

Natural alkaline particulate emissions from soils
for western Canada and all Canada¹

Emissions (10^3 tonnes·y ⁻¹)			
Region	Ca	Mg	K
British Columbia	38.8	22.7	32.3
Alberta	158	115	316
Saskatchewan	414	229	564
Manitoba	108	39.9	90.6
N.W.T./Yukon	321	126	172
Western Canada ²	719	407	1002
Canada	1100	559	1227

¹ Environmental Applications Group (1982).

² Excludes Yukon and Northwest Territories (N.W.T.), Environmental Applications Group (1982).

Table 2.7

Sea spray emissions of sulphate¹

Region	Area (km ²)	Emissions (ktonnes·y ⁻¹)
Pacific Ocean	390,000*	27,300
Atlantic Ocean	520,000*	88,400
Labrador Sea	320,000*	12,800
Hudson Bay	820,000	57,400
Beaufort Sea	50,000	3,000
Total		188,900

¹ From EPS, 1980.

* Area to 320 km limit.

highlighted by Summers (1986), who showed that:

- 1) acidity (hydrogen ion concentration) in wet deposition in western Canada CANSAP data was poorly correlated with sulphate (unlike eastern Canada and eastern U.S.).
- 2) The sum of Ca^{++} , NH_4^+ , and Mg^{++} concentrations had temporal (monthly) patterns that were similar to the sum of $\text{SO}_4^{=}$ and NO_3^- concentrations (see Figure 2.2) suggesting that sulphate and nitrate are influenced more by the soil derived Ca^{++} , or Mg^{++} , and also NH_4^+ than by SO_2 and NO_x emissions.
- 3) Relationships between calcium and sulphate concentrations in CANSAP data showed that western Canada CANSAP data (1981) were associated with a wide range of Ca^{++} concentrations, a narrow range of sulphate concentrations and increasing pH as sulphate concentration increased. In contrast, eastern Canada CANSAP data had lower pH as sulphate concentrations increased with a narrow range of calcium concentrations and a wide range of sulphate concentrations (see Figure 2.3.). Furthermore, the ratio of Ca^{++} to $\text{SO}_4^{=}$ in prairie soils is similar to that found in western Canada precipitation.

The amounts of natural particulate emissions of Ca plus Mg in eastern Canada, are small compared with western Canada (5.3×10^9 versus 7.0×10^{10} equivalents $\cdot\text{y}^{-1}$). Thus the total emissions of Ca and Mg

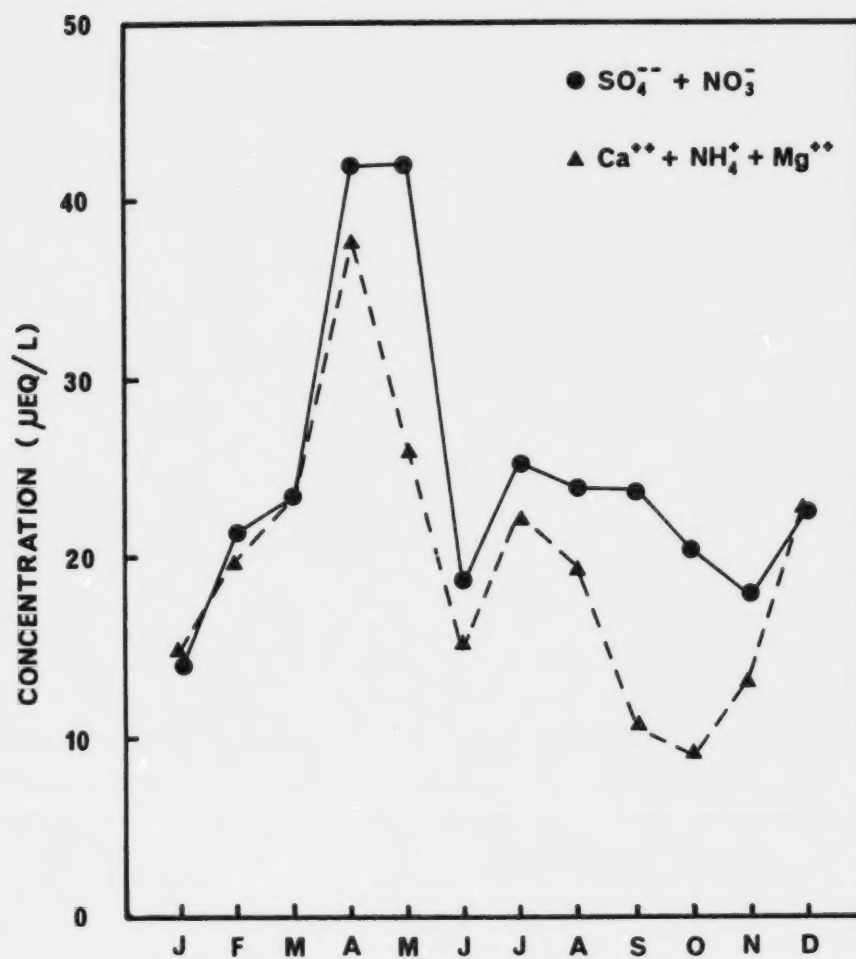


Figure 2.2: Monthly mean concentrations of selected ions at Cree Lake, July 1982 to December 1984 (from Summers 1986).

equivalents compared to SO_2 and NO_2 equivalents, are 3% in eastern Canada compared to 84% in western Canadian provinces. If size distribution is considered, the small-sized ($<10\mu\text{m}$) Ca and Mg particles in natural emissions would be only 0.16% of the total SO_2 and NO_2 emissions in eastern Canada and 5.5% in western Canada.

Thus, in eastern Canada, the sulphate loading may be used as a surrogate for acid loading. If the data in Figure 2.3 are considered, then the same will not be universally true for western Canada (i.e., sulphate may not be a surrogate for acid loading).

Sulphates

Emissions of natural sulphates arise mainly from sea spray and only in the case of British Columbia is this source significant. As a result, wet deposition data require the application of a sea salt correction in order to discount the sulphate contributed by sea spray.

In the same way that sea spray corrections are applied to wet precipitation data, it may be feasible to develop analogous corrections for the soil-derived components (e.g., calcium in wet precipitation), but the geographic variability of soil composition may be a severe complication.

2.2.5 Organics

There are several classes of organic pollutants. Emissions or ambient data are available only for volatile

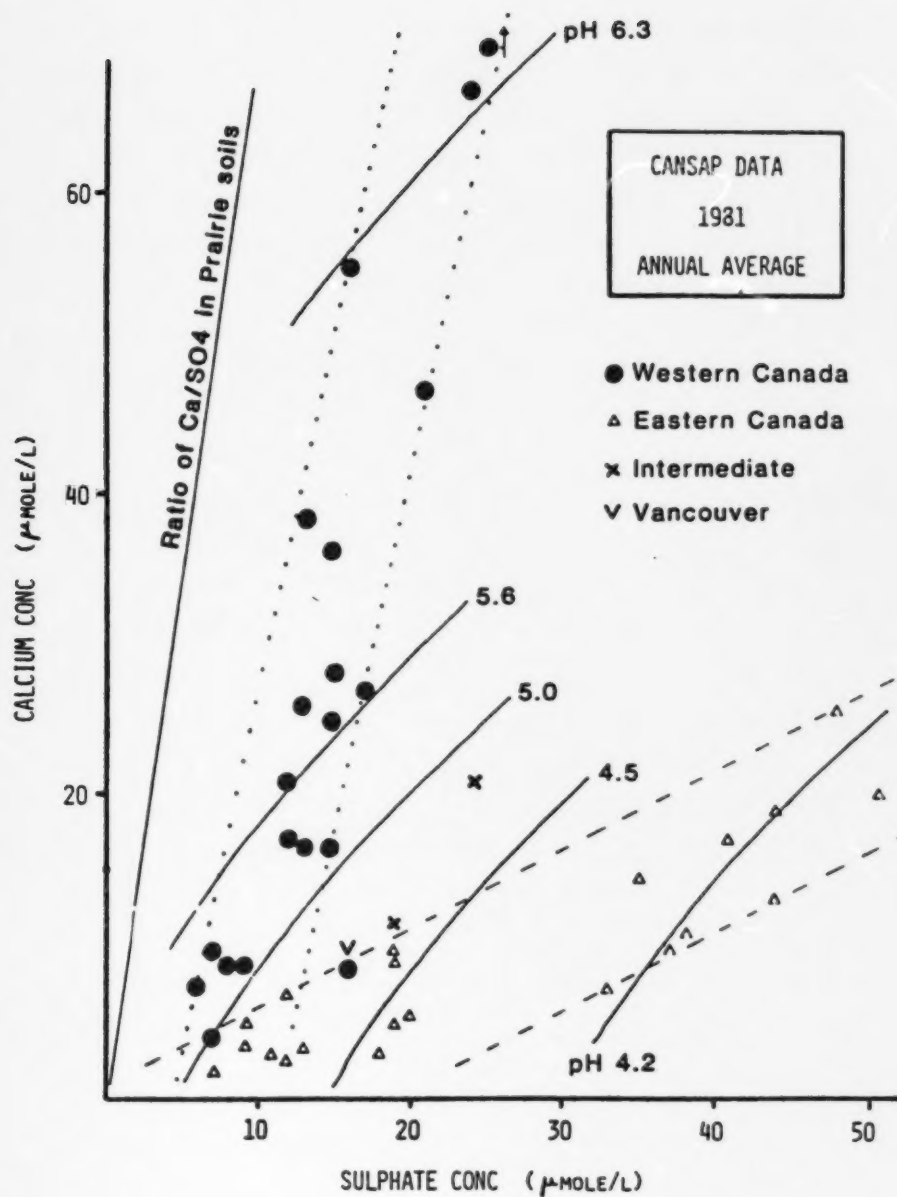


Figure 2.3: The relationship between the annual average $\text{SO}_4^{=}$ concentration, Ca^{++} concentration and pH of precipitation for the CANSAP stations in 1981. Also shown is the typical ratio of $\text{Ca}^{++}:\text{SO}_4^{=}$ in Prairie Soils.

organic compounds (VOC) (emissions data), total hydrocarbons (THC) (ambient data) or selected classes of toxic organic compounds such as pesticides and polynuclear aromatic hydrocarbons (PAH) (ambient or precipitation data). The VOC emissions and the ambient total hydrocarbons data are the most relevant in the context of target loading, since they are connected through atmospheric chemical reactions with photochemical oxidants and with organic acids in precipitation (see Figure 2.4). There is also a potential connection between oxidants and inorganic acidity, since the formation of acids from SO_2 and NO_x precursors is an oxidation process. One pathway for this process may involve oxidants (Durham and Demerjian 1984). It should be noted that direct deposition of hydrocarbons is not important for target loading since there are no direct adverse effects of hydrocarbons on receptors (with some exceptions, e.g., ethylene phytotoxicity).

The toxic organics, while important in considering human health aspects, will not be treated further since there is a general lack of emissions information and few routine measurements of ambient levels are available.

In view of the fact that the connection with photochemical oxidants and acids in precipitation is indirect, it is not worthwhile at this stage of the development of target loading to consider VOC or THC data further. For the time being it would be more appropriate to research further ozone levels (deposition) and the associated issues in western Canada and to look at available information on organic acids in western Canada precipitation.

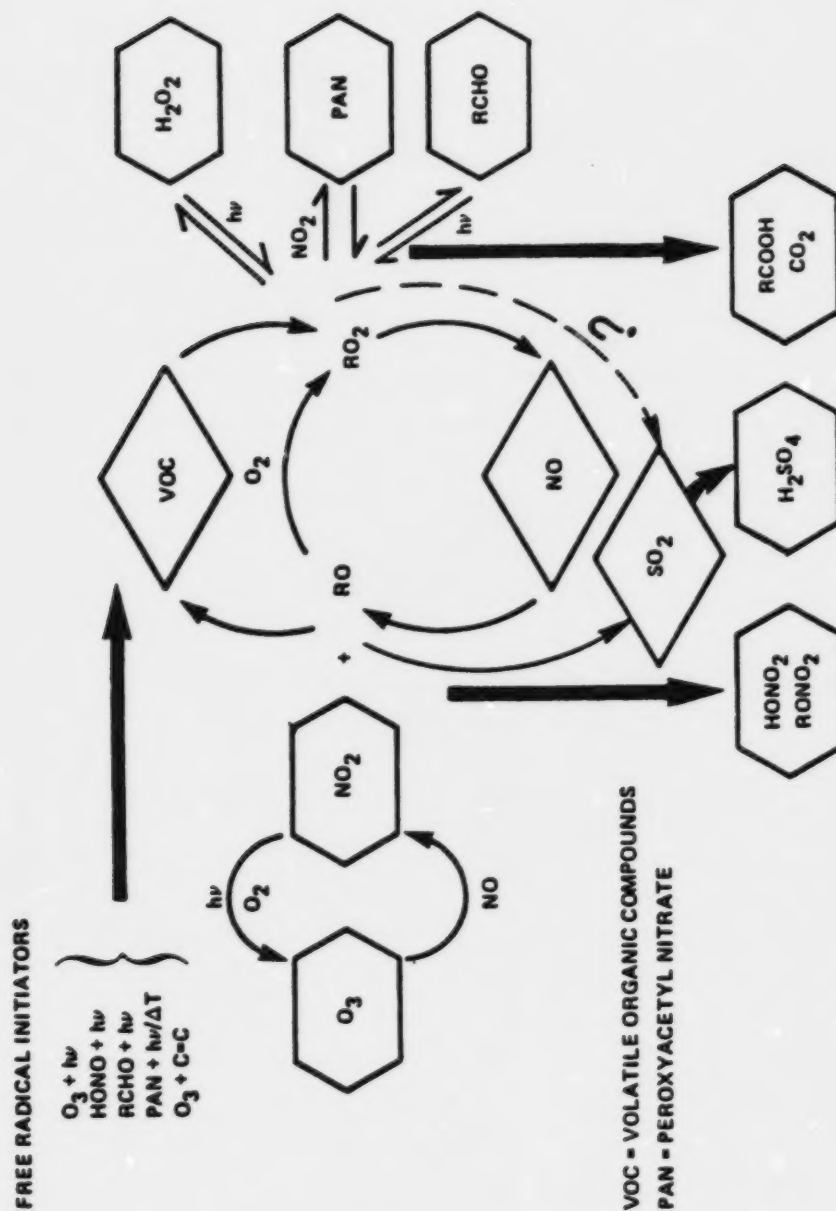


Figure 2.4: Schematic of the polluted atmospheric photo-oxidation cycle (from MOI, 1982b).

2.3 Imports and Exports

The influx of pollutants from areas outside western Canada should be examined for the potential impact from long-range transported pollutants. In addition, the knowledge of the flux of pollutants leaving the region and the levels of emission and deposition within the region will provide the basis for mass balance calculations.

Data on the fluxes entering or leaving western Canada are not well known. In contrast, information on fluxes entering or leaving eastern Canada (Galloway and Whelpdale 1980) or eastern North America (Whelpdale et al. 1984) are well known. Similar analyses for western Canada are beyond the scope of this project.

The SO_x and NO_x sources in the states of Washington, Montana, Idaho, North Dakota, and Minnesota (see Section 3, Figure 3.2) have the potential for impact through long-range transport. The western Canada LRTAP modelling study by Kociuba et al. (1984) took into account the emissions from the adjacent states in the U.S. but an overall flux from these sources was not determined. In a related study by Sandhu and Blower (1987), climatological surface and upper air wind data from Fort McMurray were cited to identify the major directions of the long-range transport of pollutants out of and within Alberta. Further study of the fluxes entering the region derived from meteorological data closer to the sources of interest (in the U.S.) would provide better estimates. In general, it is expected that imports into the region will not be significant.

Long-term estimates of the contribution of long-range transport to wet or dry deposition levels in British Columbia are not available. Analyses of the trajectories of individual storm tracks in the region suggested that transport from the state of Washington can contribute to pollutant levels in wet deposition in British Columbia (Kotturi 1984). The analysis of data from the Strait of Georgia network in conjunction with meteorological data should provide additional information.

The flux of pollutants leaving western Canada across the Manitoba-Ontario border was estimated by Whelpdale and Likens (1982) and was cited by Summers (1986) in performing rough mass balance calculations for western Canada emissions. The flux leaving western Canada by this route was ~18% relative to the total emissions (Summers 1986).

2.4 Summary

In the context of target loading, the most important pollutants are the acid-forming sulphur and nitrogen oxides and related compounds (sulphates, nitrates). In western Canadian provinces, manmade sulphur and nitrogen oxides emissions are significant percentages of the total emissions (80% and 48%, respectively). Hydrogen sulphide, which is derived mainly from natural sources, through atmospheric oxidation reactions, may also contribute to the acid loading in western Canada. The estimates of hydrogen sulphide emissions are subject to considerable uncertainty but may represent up to 23% of the total SO_2 emissions (expressing H_2S emissions as the equivalent in SO_2). Information on sulphur and nitrogen oxides emissions transported into western Canada is sparse and requires further study.

Emissions of alkaline particulate matter in western Canadian provinces were estimated to be of the order of 5% of the total acid equivalents of SO_2 and NO_x emissions, and, if present as a contaminant in precipitation samples, may have some impact on the pH levels in wet deposition in some regions of western Canada.

The ambient levels of the sulphur and nitrogen oxides in western Canada are available primarily from urban or source-specific monitoring networks. Ambient ozone monitoring data are available mainly in urban centres, but the sparse amount of rural ambient ozone data in Alberta and British Columbia suggest that further investigation of the temporal and spatial distribution of ozone levels in western Canada is warranted. Like the acid-forming nitrogen and sulphur oxides, ozone and other photochemical oxidants have the potential for causing adverse effects.

For purposes of establishing target loadings, the project's requirements were that only the acid-forming pollutants be considered. However, it is clear that suspended particulate matter, and ammonia are also important since they have potential for influencing the acidity of precipitation in western Canada. In addition, it should be noted that ozone may be a pollutant of concern in western Canada, but the application of the target loading concept to ozone would be challenging.

3.0 ADEQUACY OF EXISTING DATABASES FOR ESTIMATING LOADINGS OF ACID FORMING POLLUTANTS

The estimation of loadings for the acid-forming pollutants requires knowledge of the wet and dry components of deposition. The latter has been estimated most often by modelling studies which calculate the concentration and the loading (the product of the concentration and deposition velocity). The former is most often determined from measured wet deposition, although models may be used to provide such estimates. Both methods will need to be applied in western Canada. The types of information required for these methods (i.e., model inputs and monitoring network requirements) are first identified, and then the extent to which these data (i.e., the databases) are available for western Canada is discussed. The adequacy of the existing information will be evaluated bearing in mind proposed enhancements to the databases. The adequacy of the databases for target loading will be assessed by first identifying the attributes that are appropriate for evaluation, and then making the assessment to an appropriate level of detail. Given the limited scope of this project, the assessment must be qualitative.

3.1 Information Requirements for Estimating Target Loadings by Modelling

3.1.1 Model Inputs

The Western Canada LRTAP Technical Committee identified the need for urban photochemical, mesoscale deposition and advanced complex deposition and oxidant models (LRTAP Workshop No. 4. 1985) as well as statistical models (Western Mesoscale Modelling Task Group 1985). Models selected for use in western Canada are listed in Table 3.1.

Table 3.1

Models selected by Western Canada LRTAP
Technical Committee for use in western Canada

Application Type	Model
Urban photochemical	PLMSTAR AIRSHED
Mesoscale deposition	MESOPUFF II RTM III
Advanced complex	ADOM
Statistical	OME-STATMOD FISHER RCDM

Of relevance to target loading are the statistical and the mesoscale deposition models. The input requirements for these models are discussed below.

Currently, there are three statistical models which are at varying stages of development for application by the Western Canada LRTAP Technical Committee to western Canada. The three models are the Regional Climatological Deposition Model (RCDM), the Ontario Ministry of the Environment model (OME-STATMOD), and an enhancement of Fisher's statistical model (SERTAD). All three models have been implemented by the Atmospheric Environment Service on their computer at Downsview (CSC 1986).

The models require inputs that were categorized into four types, namely meteorological, emission, physical and chemical parameterization, and diffusion parameter data. These input types for the statistical models are summarized in Appendix IV.

It should be noted that the time resolution of the Lagrangian and Eulerian models (hourly) and hence the current costs associated with running these models (episode models), make them inappropriate for target loading applications.

3.1.2 Adequacy of Model Inputs

Meteorology and Diffusion Data

In general, the required meteorological data are available, or derivable, from the meteorological archives maintained by the Atmospheric Environment Service in Canada,

and by the National Weather Center in the United States. However, the following areas of difficulty should be noted:

- 1) The wind observing network is rather sparse, particularly for the upper levels. It is therefore necessary to interpolate these data in both space and time, noting that best results will be obtained if the interpolation routines take into account the physical properties of the atmosphere. Such an interpolation routine is normally considered part of the model. This comment applies to the Lagrangian and Eulerian models.
- 2) Also of concern in the case of the Lagrangian and Eulerian models is the volume of data required. This is a consequence of the spatial and temporal resolution employed in these models.
- 3) Eddy diffusivities are normally calculated in the more complex models, and specified in the simpler models, since they are not available as observed data. Tests on SERTAD indicated that the model predictions are relatively insensitive to the value of the vertical diffusivity.

Chemical and Physical Parameter Values

All of the models require values for chemical and physical parameters. These requirements are summarized in Appendix IV.3, and are naturally more extensive for the more complex models. The relevant parameters include wet and dry scavenging rates and chemical reaction rates, and normally

constitute a part of the model, either being inserted as numerical values, or calculated using other data. An example of the latter case is the calculation of deposition velocities in ADOM as functions of atmospheric conditions, ground cover, and surface roughness.

Also included in Appendix IV.3 are model requirements for physical data including ground cover and surface roughness, as noted above, as well as topographic data (i.e., elevation). These data are available from provincial, state and federal geological surveys, and other sources.

To summarize, the necessary meteorology and dispersion data are generally available. However, presentation of the data in the appropriate format for Lagrangian and Eulerian models may require a substantial amount of work.

3.2 Monitoring Requirements for Estimating Loadings

The estimation of loadings for acid-forming pollutants requires monitoring data that will provide representative indicators of ambient air quality levels and precipitation quality and quantity throughout the region. For ambient air quality data, the concentrations of gaseous and particulate sulphur and nitrogen compounds are required (i.e., gaseous SO_2 , NO_2 and NO_x , particulate $\text{SO}_4^{=}$, NO_3^- , Ca^{++} , Mg^{++} , Cl^- , NH_4^+); while in the case of precipitation data, the amount of precipitation and the concentrations of various substances ($\text{SO}_4^{=}$, NO_3^- , H^+ , Ca^{++} , Mg^{++} , Cl^- , NH_4^+) are required.

3.2.1 Air Quality Monitoring Networks

The air quality monitoring networks in western Canada

are listed in Table 3.2. Included in the table are the operating agency, the area monitored, the number of stations and a selected list of parameters measured. The parameters are selected from the list of gaseous (SO_2 , NO_2 , NO_x , or O_3) and particulate (TSP , $\text{SO}_4^{=}$, NO_3^{-}) pollutants.

The major networks are those operated by the federal (National Air Pollution Surveillance Program) and provincial agencies. Most of these stations are located in urban centres. In the case of Alberta, there is an extensive 'network' of monitoring stations located in the environs of industrial facilities for petroleum processing and power generation. These stations are operated by industry but are usually required to provide ambient monitoring data for specific pollutants as one of the requirements of the provincial permit to operate. In British Columbia, industrial permits require that monitors for the detection of H_2S , or SO_2 (or both) and TSP be operated at mining or pulp and paper industrial sites. Similar, but much less extensive industry operated networks with air quality monitoring stations exist in Manitoba and Saskatchewan.

3.2.2 Wet Deposition Networks

Wet deposition networks in operation in western Canada are listed in Table 3.3 along with the current number of stations and parameters monitored. The locations of these stations are indicated in Figure 3.1.

TABLE 3.2
Air quality monitoring networks in western Canada

Network	Agency	Area Monitored	Number of Stations	Variables, and (Number of Monitors)
National Air Pollution Surveillance Network (NAPS)	Environmental Protection Service, Environment Canada	Canada (urban centres)	39 ¹	SO ₂ (15), NO ₂ (15), O ₃ (15), TSP(34)
Manitoba Ambient Air and Precipitation Quality Monitoring Network	Environmental Management, Environment and Workplace Safety and Health	Manitoba	10	SO ₂ (5) ² , NO ₂ (3) ² , O ₃ (3) ² , SO ₄ (6) ³ , SO ₄ (6) ³ , NO ₃ (6) ³ , TSP(6) ³
Saskatchewan	Air Pollution Control Branch, Saskatchewan Environment	Saskatchewan (urban areas)	6	SO ₂ (4) ⁴ , NO _x (2) ⁴ , O ₃ (2) ⁴ , TSP(7) ⁴
Alberta Environment Air Monitoring Network	Air Quality Control Branch, Alberta Environment	Alberta (urban areas)	6	SO ₂ (2) ⁶ , NO _x (6) ⁷ , O ₃ (6) ⁷ , TSP(6) ⁸
Alberta Exposure Station Program	Air Quality Control Branch* (Alberta Env.)	Alberta	2112	Total sulphation (2112), H ₂ S (1276)
British Columbia Ministry of the Environment (BC MOE)	BC MOE	British Columbia	26	SO ₂ (10), NO ₂ (5), O ₃ (9), H ₂ S(15)

TABLE 3.2 (continued)
Air quality monitoring networks in western Canada

Network	Agency	Area Monitored	Number of Stations	Variables, and (Number of Monitors)
BC MOE Permittee Network	BC MOE Permittees	British Columbia	74	SO ₂ (20), H ₂ S(15), TSP(51)
CAPMoN	Atmospheric Environment Service, Downsview	Canada	3 ¹	Particulate SO ₄ ⁼ , NO ₃ ⁻ , Cl ⁻ , H ₄ ⁺ , Na ⁺ , K ⁺ , Gaseous NHO ₃ , SO ₂
Greater Vancouver Regional District (GVRD)	GVRD Pollution Control	GVRD	36	SO ₂ (8), NO ₂ (12), O ₃ (13), TSP(36)
Sour Gas Industry	-	Sour gas plants, Alberta	91	SO ₂ (90), H ₂ S(86)
Refineries	-	Alberta	4	SO ₂ (5), NO _x (4), TSP(6)
Pulp and Paper	-	Alberta	2	SO ₂ (2), H ₂ S(2)
Oil Sands	-	Alberta	10	SO ₂
Power Plants	-	Alberta	-	SO ₂ (8), NO ₂ (12), NO _x (16), TSP (11)

TABLE 3.2 (continued)
Air quality monitoring networks in western Canada

Network	Agency	Area Monitored	Number of Stations	Variables and (Number of Monitors)
Chemical Plants	-	Alberta	9	-
Heavy Oil	-	Alberta	15	-

¹ Sites in Western Canada only

² Includes 2 NAPS stations

³ Includes 4 NAPS stations

⁴ Includes 2 NAPS stations

⁵ Includes 8 NAPS stations

⁶ Includes 2 NAPS stations

⁷ Includes 4 NAPS stations

⁸ Includes 6 NAPS stations

* The Alberta Environment Exposure Station program is operated by the Air Quality Control Branch; however, industries in this province have several hundred networks of various sizes as a requirement of various Licenses. These networks are additional to those operated by the Province.

- No Information, or Not Applicable.

Table 3.3: Summary of wet deposition networks in western Canada

Network	Agency	Number of Stations/Area Monitored	Sampling Interval/Start Date	Variables
Canadian Network for Sampling Precipitation (CANSAP)	Atmospheric Environment Service, Environment Canada	28 Western Canada	Monthly January 1977 to 1982	pH (lab), conductivity, SO_4^{2-} , NO_3^- , Cl^- , NH_4^+ , Na, Ca ⁺⁺ , K ⁺ , Mg
Canadian Air and Precipitation Monitoring Network (CAPMON)	Atmospheric Environment Service, Environment Canada	3 Western Canada	Daily September 1983	pH (lab), conductivity, SO_4^{2-} , NO_3^- , Cl^- , NH_4^+ , Na, Ca ⁺⁺ , K ⁺ , Mg
Canadian Network for Sampling Organic Compounds in Precipitation (CANSOC)	Water Quality Branch, Inland Waters Directorate, Environment Canada	Canada 12	Monthly Mid. 1977	lindane, heptachloropoxide, aldrin, dieldrin, hexachlorobenzene, heptachlor, α BHC, p,p'-DDE, p,p'-DDT, o,p'-DDT, mirex, endrin, p,p'-methoxychlor, a-endosulfan, b-endosulfan, cis-chlordane, trans-chlordane, Aroclor, total (PCB)
South Coast, B.C. Network	Atmospheric Environment Service, Environment Canada	13 West Vancouver Island to Agassiz along U.S. Canada border	1983	pH, acidity, SO_4^{2-} , NO_3^- , Cl^- , Na ⁺⁺ , K ⁺ , Ca ⁺⁺ , Mg, NH_4^+
B.C. Lower Mainland Network	Atmospheric Environment Service, Environment Canada	7 Georgia Strait, B.C. 16	1980/81	pH, acidity, SO_4^{2-} , NO_3^- , Cl^- , Na ⁺⁺ , K ⁺ , Ca ⁺⁺ , Mg ⁺⁺ , NH_4^+
Acid Precipitation Monitoring Program in British Columbia	Waste Management Branch, Ministry of the Environment, British Columbia	6 British Columbia	Weekly 1983	pH (lab), total acidity, acidity (free), SO_4^{2-} , NO_3^- , Cl^- , NH_4^+ , Na ⁺⁺ , K ⁺ , Mg ⁺⁺ , Ca ⁺⁺

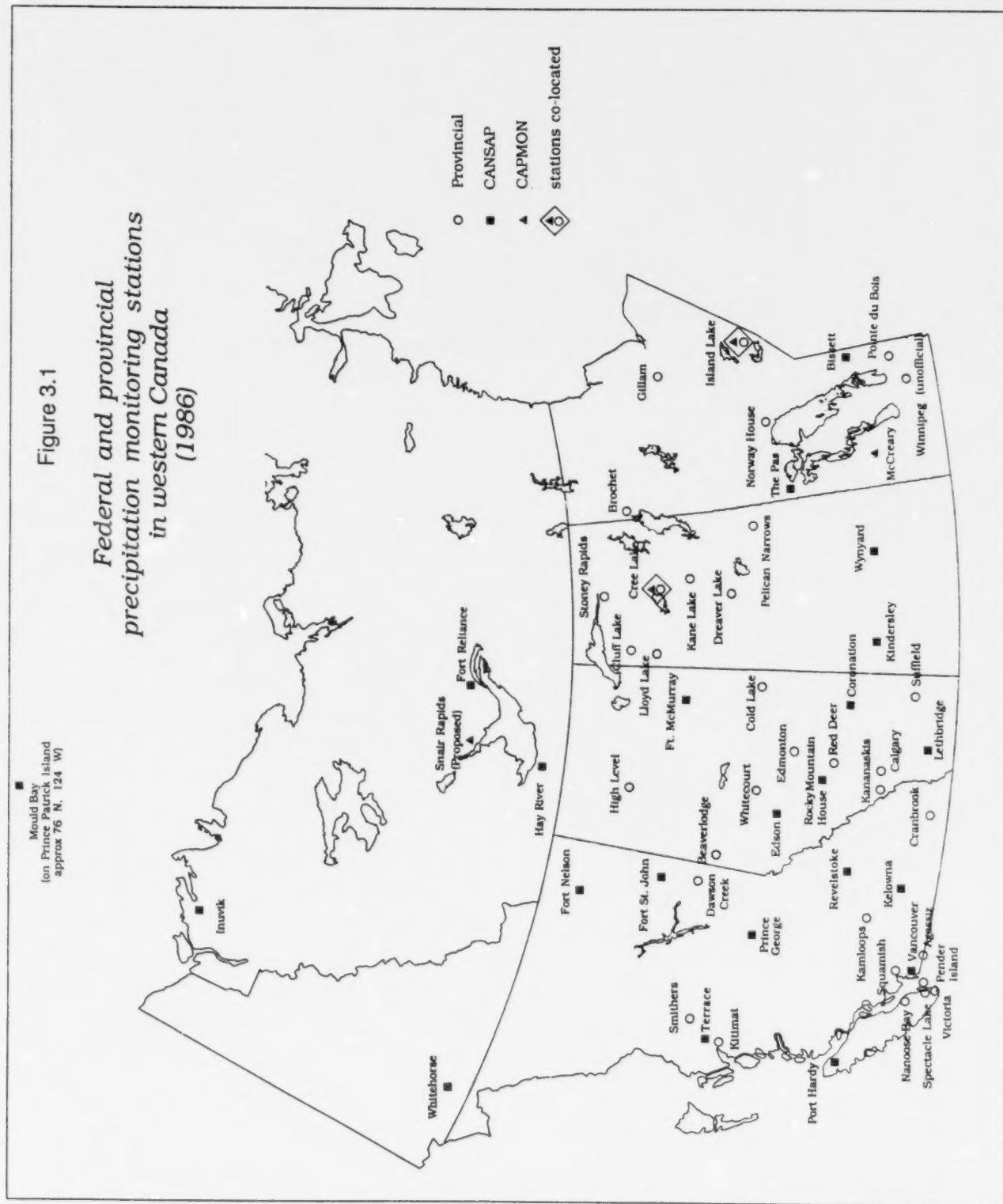
Table 3.3 (cont'd): Summary of wet deposition networks in western Canada

Network	Agency	Number of Stations/Area Monitored	Sampling Interval/Start Date	Variables
Precipitation Quality Monitoring Program (PQMP)	Air Quality Control Branch, Pollution Control Division, Alberta Environment	10 Alberta*	Monthly April 1978	pH, conductance, acidity/ALK, SO_4^{--} , NO_3^- , Cl^- , NH_4^+ , K^+ , Ca^{++} , Mg^{++} , Na^+
Precipitation Chemistry in the Oil Sands Area of Northern Alberta Program (AQSERP)	Research Management Division, Alberta Environment	14 Oil Sands Area of Northern Alberta	Event (summer only) summers of 1976-1979	pH (lab), acidity, conductivity, SO_4^{--} , NO_3^- , Cl^- , NH_4^+ , Na^+ , Ca^{++} , K^+ , Mg^{++}
Saskatchewan Precipitation Monitoring Network	Air Pollution Control Branch, Environment Saskatchewan	6 Northern Sask. + 1 collocated at Cree Lake	Event, July 1982	pH (lab), acidity, SO_4^{--} , NO_3^- , NH_3 , Ca^{++} , Cl^- , K^+ , Mg^{++} , Na^+ , 26 Trace Metals
Manitoba Network for Precipitation Collection (MNPC)	Environmental Management Division, Manitoba Department of Environment and Workplace Safety and Health	6 North-Eastern Manitoba	Daily October 1980	pH (lab), acidity, SO_4^{--} , NO_3^- , Cl^- , NH_4^+ , Ca^{++} , Na^+ , K^+ , Mg^{++}
Manitoba Snow Monitoring Network		39	3/yr 1983	pH, SO_4^{--} , metals
B.C. Provincial Precipitation Network	British Columbia Ministry of the Environment	7	Weekly 1982	pH, Ca^{++} , M^{++} , NH_4^+ , Na^+ , Cl^- , SO_4^{--} , NO_3^- conductance
Saskatchewan Power Corporation Air and Precipitation Study		1	1983	pH, SO_4^{--} , NO_3^-

* PQMP (Alberta) had 9 stations in 1985. In 1987 one more station was added, raising the total to 10.

Figure 3.1

*Federal and provincial
precipitation monitoring stations
in western Canada
(1986)*



3.2.3 Adequacy of Monitoring Database

In the case of ambient air quality requirements for estimating loadings, the air quality database should provide data that are:

- 1) regionally representative;
- 2) have adequate temporal resolution;
- 3) complete in terms of the range of pollutants monitored and the period of record;
- 4) reliable and accurate; and
- 5) consistent in quality assurance and quality control (QA/QC) and reporting.

The air quality database provides good coverage of urban centres and point-source oriented indications of air quality. These sites do not provide regionally representative air quality data of the type needed for estimating loadings.

In the case of the point-source related monitoring networks in Alberta, especially, and British Columbia, the sites do include background stations which may provide regionally representative air quality data. The available SO₂ and NO₂ data from background sites in Alberta, British Columbia, and Saskatchewan bear further investigation regarding the accuracy, reliability and consistency, since different institutions run the instruments and no consistent QA/QC protocols or database system are in place. In view of this, the data from such stations need to be viewed as inadequate, but bear more detailed evaluation.

The range of pollutants required to establish target loadings for acidic deposition includes routine gaseous

sulphur and nitrogen oxides, and particulate $\text{SO}_4^{=}$, NO_3^- , Ca^{++} , Mg^{++} , and NH_4^+ . Information on gaseous nitric acid, at least on a periodic or special study basis, would provide complementary information.

The precipitation database is derived mainly from CANSAP sites, established or proposed CAPMoN sites, and provincial networks. The regional representativeness of the network may be appropriate, but large gaps occur in northern British Columbia and southern Saskatchewan. The coverage of the Yukon and Northwest Territories is very limited.

The period of record dates from 1977, but analyses of the data resulted in recommendations for improved siting and instrumentation and changes in the sampling frequency (Barrie and Sirois 1982). The CANSAP network in western Canada has been incorporated into the CAPMoN network (Lechner, pers. comm., Saskatchewan Environment 1987).

The provincial networks all use a variety of sampling equipment, have sampling frequencies ranging from daily to monthly, and appear to have varying quality assurance and quality control protocols. Clearly, the data from these networks are not consistent in terms of temporal resolution and are not likely to be of consistent reliability.

3.3 Adequacy of Emission Inventories

The adequacy of an emissions database is determined by the need for various kinds of information. For example, mathematical models for transport and deposition rely heavily on emission inventories. Episodic models, however, need emission inventories and the characteristics of these

emissions (e.g., stack height, temperature, and exit speed). Notwithstanding the particular needs of episodic models, the desirable attributes of emission inventories are as follows:

- completeness (with respect to the range of chemical species inventoried);
- accuracy (includes the accuracy of treatment of individual sources or source sectors, and the completeness of inclusion of all sources);
- spatial resolution;
- temporal resolution;
- chemical resolution (many models require separation of NO_x to NO and NO_2 and the allocation of organic species to reactivity classes); and
- currency.

The specific requirements of selected models with respect to these attributes are summarized in Table 3.4. It can be seen that the requirements of SERTAD and MESOPUFF II are met by the NEIS and MOI inventories, except that improved temporal resolution is required to exploit to the full the capabilities of MESOPUFF II.

The more sophisticated emissions inventory requirements of ADOM are clear from Table 3.4. These will be met by inventories constructed to the standards employed in the NAPAP emission inventory.

The importance of providing an estimate of the uncertainty in the emissions has been recognized (e.g., MOI, 1982a; Benkovitz 1985). The MOI estimates of precision are given in Table 3.5, noting that the precision may be

Table 3.4
Summary of emission inventory requirements for selected models

Attribute	Statistical SERTAD	Model Lagrangian MESOPUFF II	Eulerian ADOM
Completeness	SO ₂	SO ₂ , NO _x	SO ₂ , NO _x , VOC, CO, NH ₃
Accuracy	See Footnote ¹	See Footnote ¹	See Footnote ¹
Spatial resolution	127 km x 127 km	127 km x 127 m	127 km x 127 km
Temporal resolution	Season, year	1 hr	3 hrs
Chemical resolution	Not applicable	NO, NO ₂	NO, NO ₂ , ten VOC classes
Currency	See Footnote ²	See Footnote ²	See Footnote ²

¹ The best possible accuracy should be used, since this limits the accuracy of the model predictions.

² Must match the situation being modelled.

Table 3.5
Precision estimates for emissions data¹

Region	Precision (%)	
	SO ₂	NO _x
British Columbia/Yukon Northwest Territories	17	23
Alberta	29	29
Saskatchewan	42	25
Manitoba	10	22

¹ From MOI, 1982a.

substantially better than the accuracy. The natural emissions are stated to be order of magnitude estimates (EPS 1980, 1981).

3.4 Existing Loadings of Acid-Forming Pollutants in Western Canada

In the remainder of this section, we will present a synopsis of the information on the available measurements of pollutant loadings (wet and dry deposition of $\text{SO}_4^{=}$, NO_3^- , and H^+ especially) in western Canada. In addition, the published information on model estimates of regional scale deposition will be summarized. These data, when viewed along with the dose-response information for appropriate pollutant/receptor systems, will allow assessment of the potential for damage and the establishment of suitable target loadings in order to protect chosen receptors.

3.4.1 Loadings in Western Canada

Estimates of the amounts of acid-forming pollutants deposited in western Canada may be derived from federal and provincial precipitation networks (wet deposition), from more limited representative air quality data (dry deposition), or from modelling studies.

Wet Deposition

Regional Estimates

In order to estimate the gross deposition in western Canada, stations remote from sources are required. Recently, Summers (1986) suggested that data from the CAPMoN station at

Cree Lake were most reliable and most representative of deposition in the Canadian Shield area across the northern Prairie provinces.

Summers (1986) estimated sulphate loadings in wet and dry deposition of 3.6 and $1.5 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ respectively for Cree Lake. Wet sulphate deposition showed strong seasonal variation ranging from the equivalent of $\sim 0.009 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{mo}^{-1}$ in January to March to $0.07 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{mo}^{-1}$ in May. Dry deposition, which showed no strong seasonal variation, ranged from 0.008 to $0.02 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{mo}^{-1}$. These estimates indicate that wet deposition is 2.4 times greater than dry deposition. [Summers also noted the special problems in the sampling of precipitation in the relatively dusty (high TSP levels) agricultural Prairie regions.]

Summers estimated that the integrated wet deposition of sulphur (S) in the four western provinces is equal to the manmade sulphur (S) emissions and that background wet SO_4 deposition at Cree Lake ($\sim 1.0 \text{ kg SO}_4 \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$) accounted for $\sim 50\%$ of the total.

Estimates of regional deposition patterns have been made based on data collected in the CANSAP and NADP networks (Semonin 1982) and for the CANSAP, APN, NADP and MAP3S networks (Barrie and Hales 1984). The NADP data, which are collected on a weekly basis, were volume weighted to a monthly basis in order to be comparable with CANSAP data which has a one-month collection period.

Maps showing the annual precipitation-amount-weighted concentration (1980 data) patterns for North America for

hydrogen ion, pH, sulphate, nitrate, and ammonium ions are shown in Figures 3.2 to 3.6 (Barrie and Hales 1984). The annual deposition patterns for sulphate, nitrate, hydrogen, and ammonium ions are shown in Figures 3.7 to 3.10.

The lowest pH levels in western Canada were about 5, in contrast to pH levels of 4.1-4.2 in eastern United States. Barrie and Hales (1984) reported that based on ion-balance analyses, high concentrations of ions in western Canada precipitation are associated with non-acidic materials--in contrast to eastern North America where most sulphates originate from acidic compounds. Similarly, the high nitrate levels in western Canada are associated with wind-blown dust as indicated by low acidity levels and high calcium levels. Ammonium concentration patterns are different from those of hydrogen, sulphate, and nitrate. Ammonium levels in western Canada range from 0.5 to 21 $\mu\text{mol}\cdot\text{L}^{-1}$. High concentrations of ammonium ion typically associated with agricultural activities are not evident in western Canada.

The deposition patterns (Figures 3.7 to 3.10) are similar to the corresponding concentration patterns but are subject to uncertainties due to the method used to determine isopleths (Barrie and Hales 1984). The annual sulphate loadings in western Canada ranged from 1.4 to 24 $\mu\text{mol}\cdot\text{m}^{-2}$ (1.3 to 23.1 $\text{kg}\cdot\text{ha}^{-1}$).

Similar estimates of sulphate wet deposition based on a five year (1978 to 1982) summary of data from Alberta provincial stations and selected stations in Manitoba, Saskatchewan, and British Columbia were compiled by Lau et al. (1985). The five-year mean wet deposition loadings in

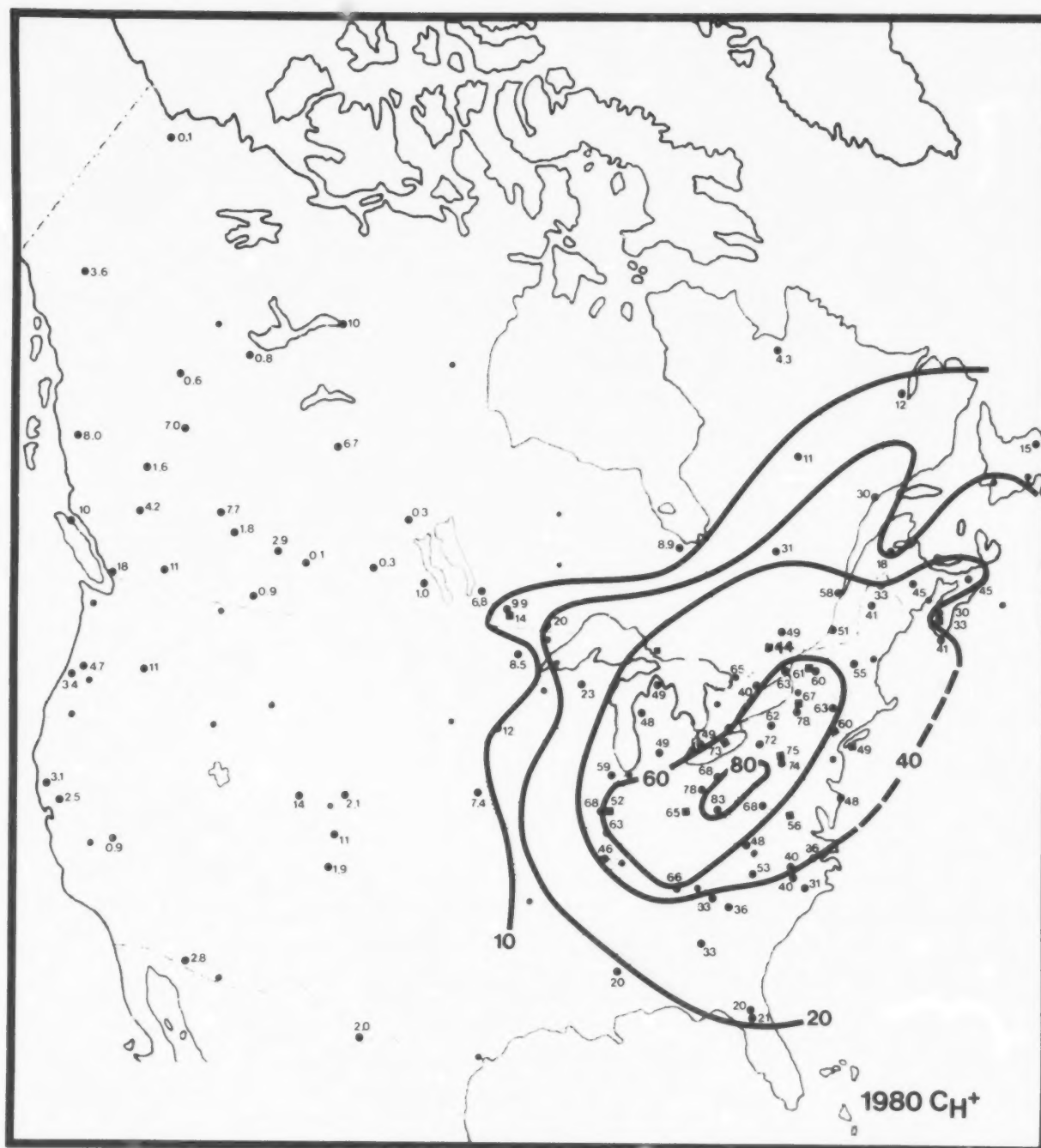


Figure 3.2: The spatial distribution of the precipitation-amount-weighted annual mean hydrogen ion concentration ($\mu\text{mole}\cdot\text{L}^{-1}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).



Figure 3.3: The spatial distribution of the precipitation-amount-weighted annual mean hydrogen ion concentration expressed as pH in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

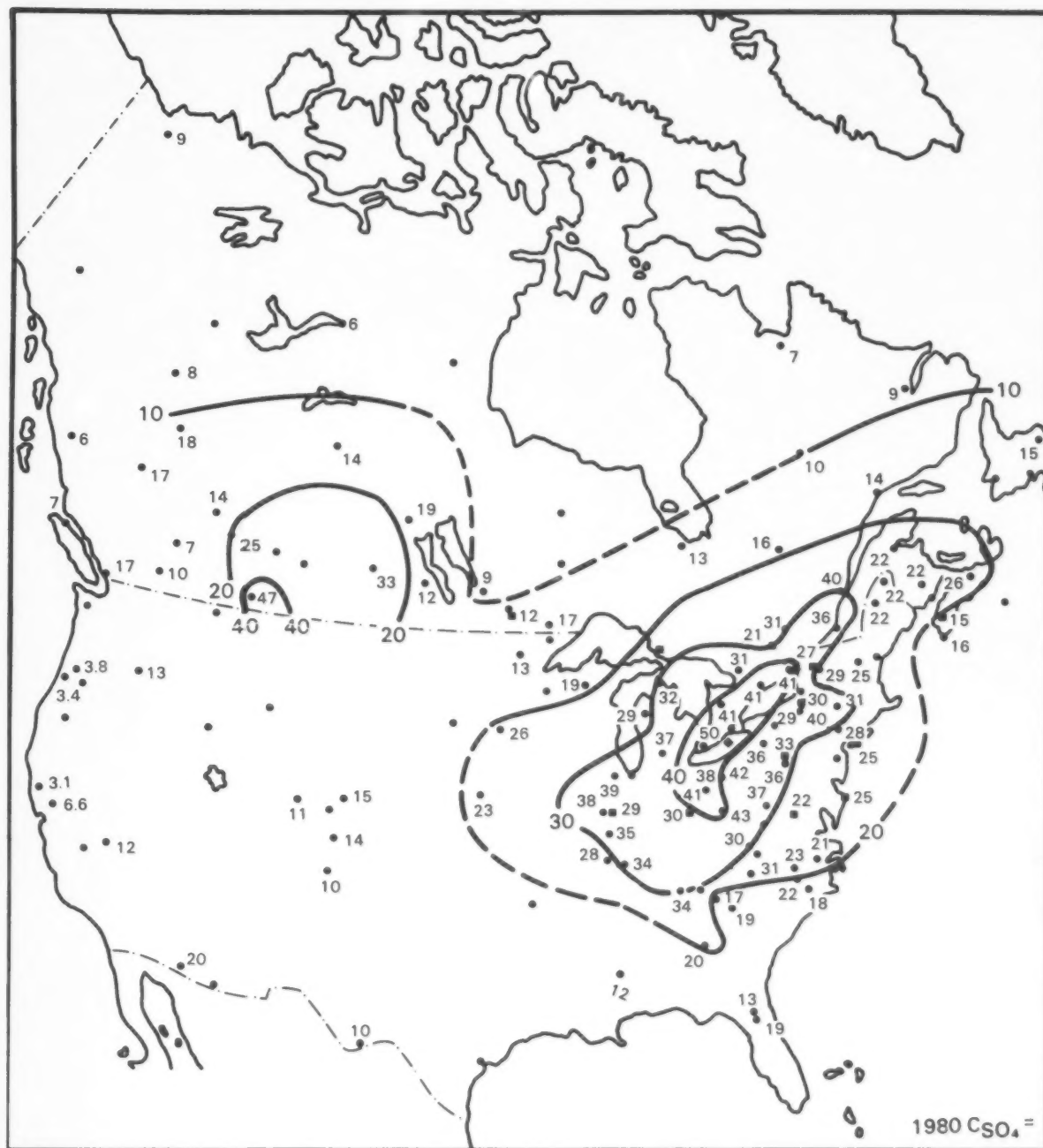


Figure 3.4: The spatial distribution of the precipitation-amount-weighted annual mean sulphate ion concentration ($\mu\text{mole}\cdot\text{L}^{-1}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

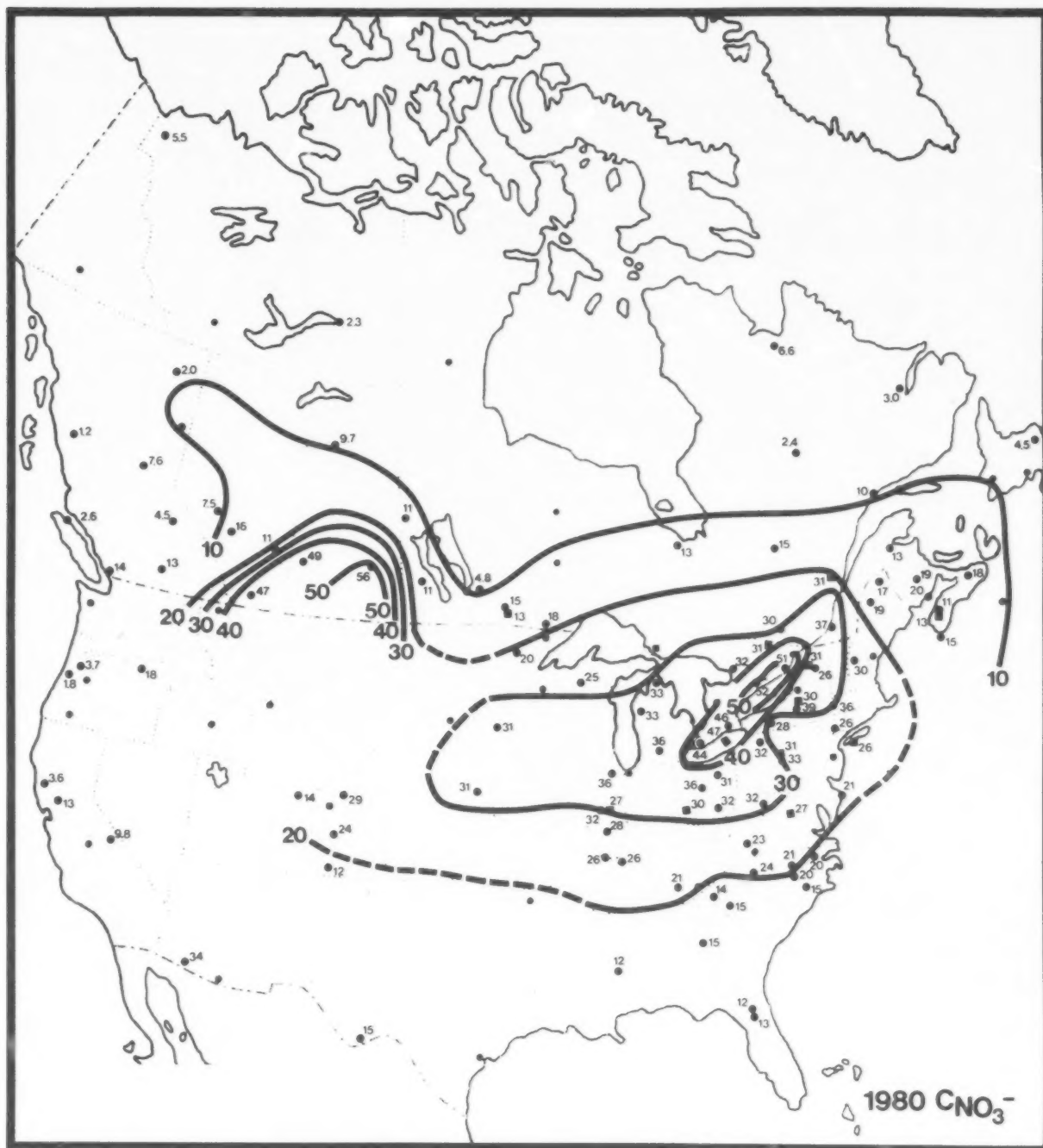


Figure 3.5: The spatial distribution of the precipitation-amount-weighted annual mean nitrate ion concentration ($\mu\text{mole} \cdot \text{L}^{-1}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

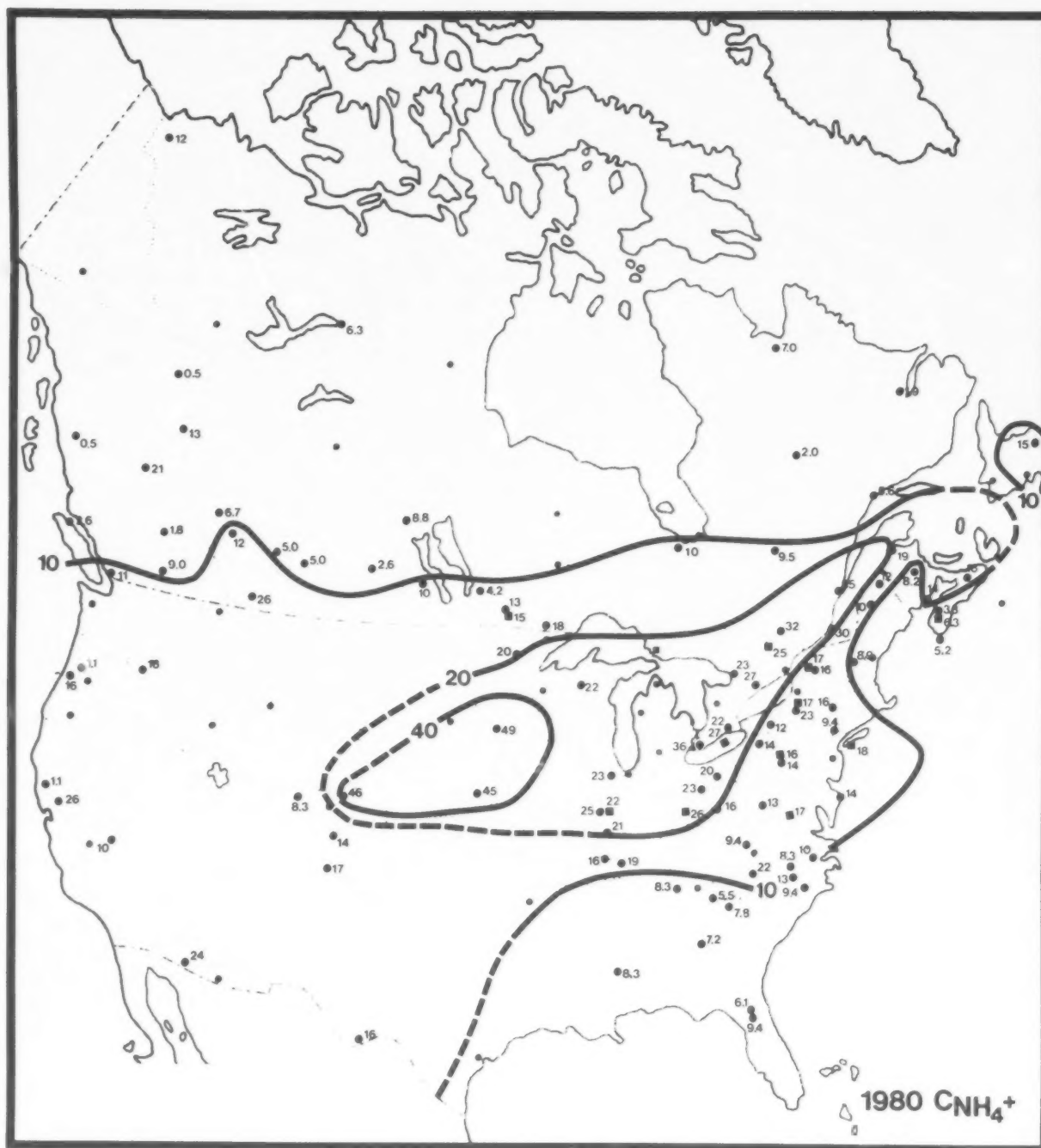


Figure 3.6: The spatial distribution of annual wet deposition of ammonium ion concentration ($\text{mmole} \cdot \text{L}^{-1}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).



Figure 3.7: The spatial distribution of annual wet deposition of sulphate ion ($\text{mmole} \cdot \text{m}^{-2}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

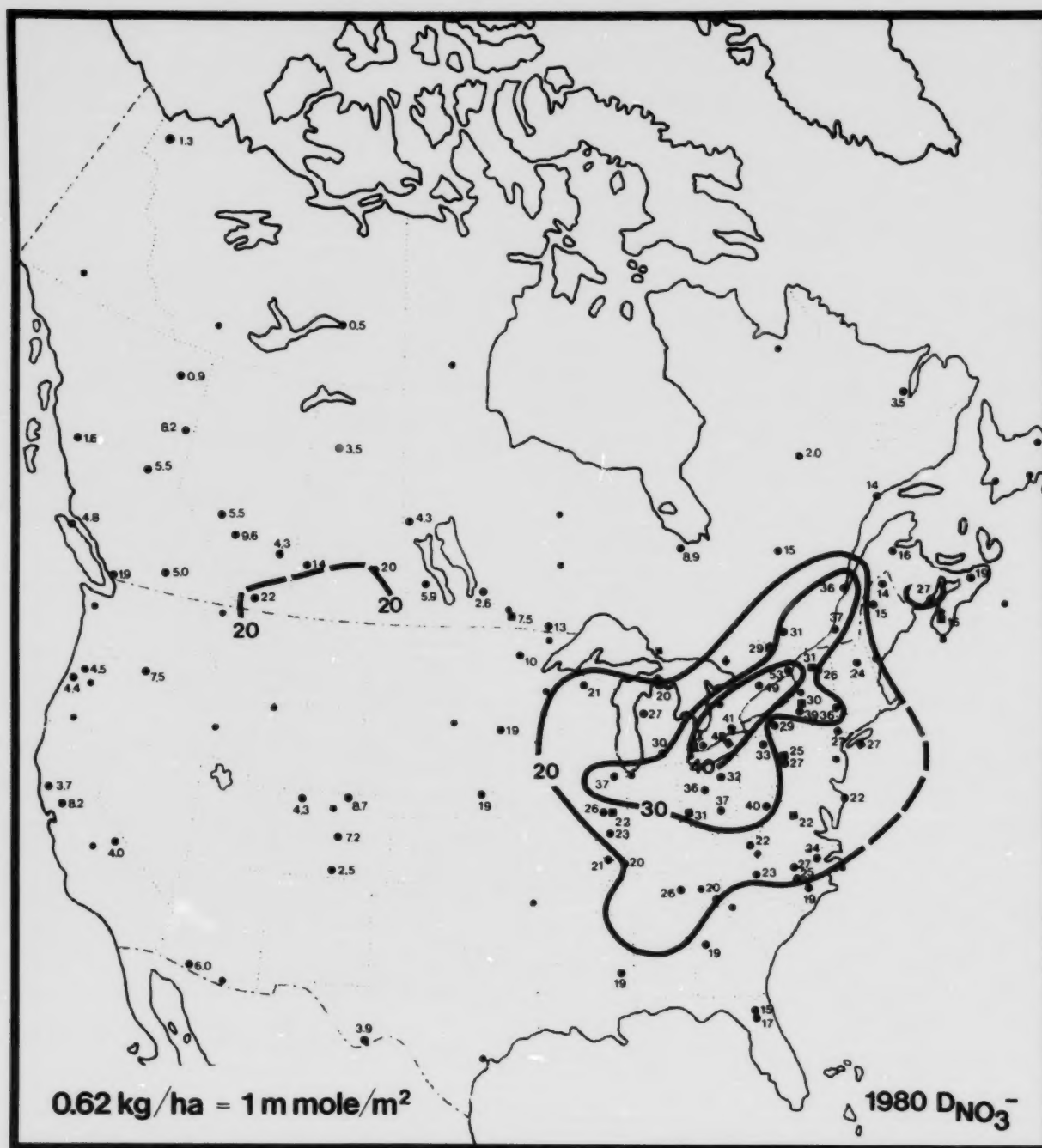


Figure 3.8: The spatial distribution of annual wet deposition of nitrate ion (mmole·m⁻²) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

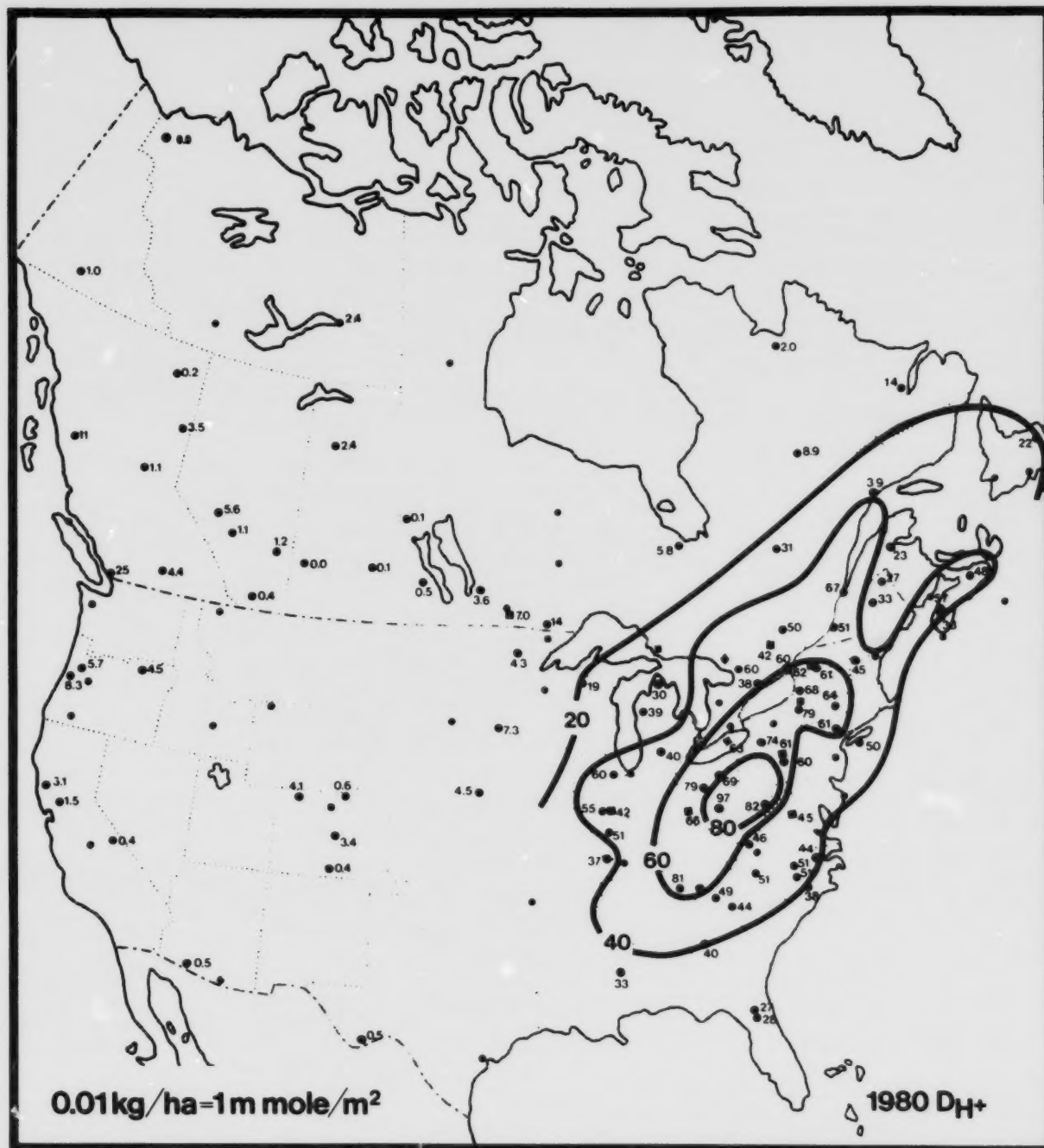


Figure 3.9: The spatial distribution of annual wet deposition of hydrogen ion ($\text{mmole} \cdot \text{m}^{-2}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

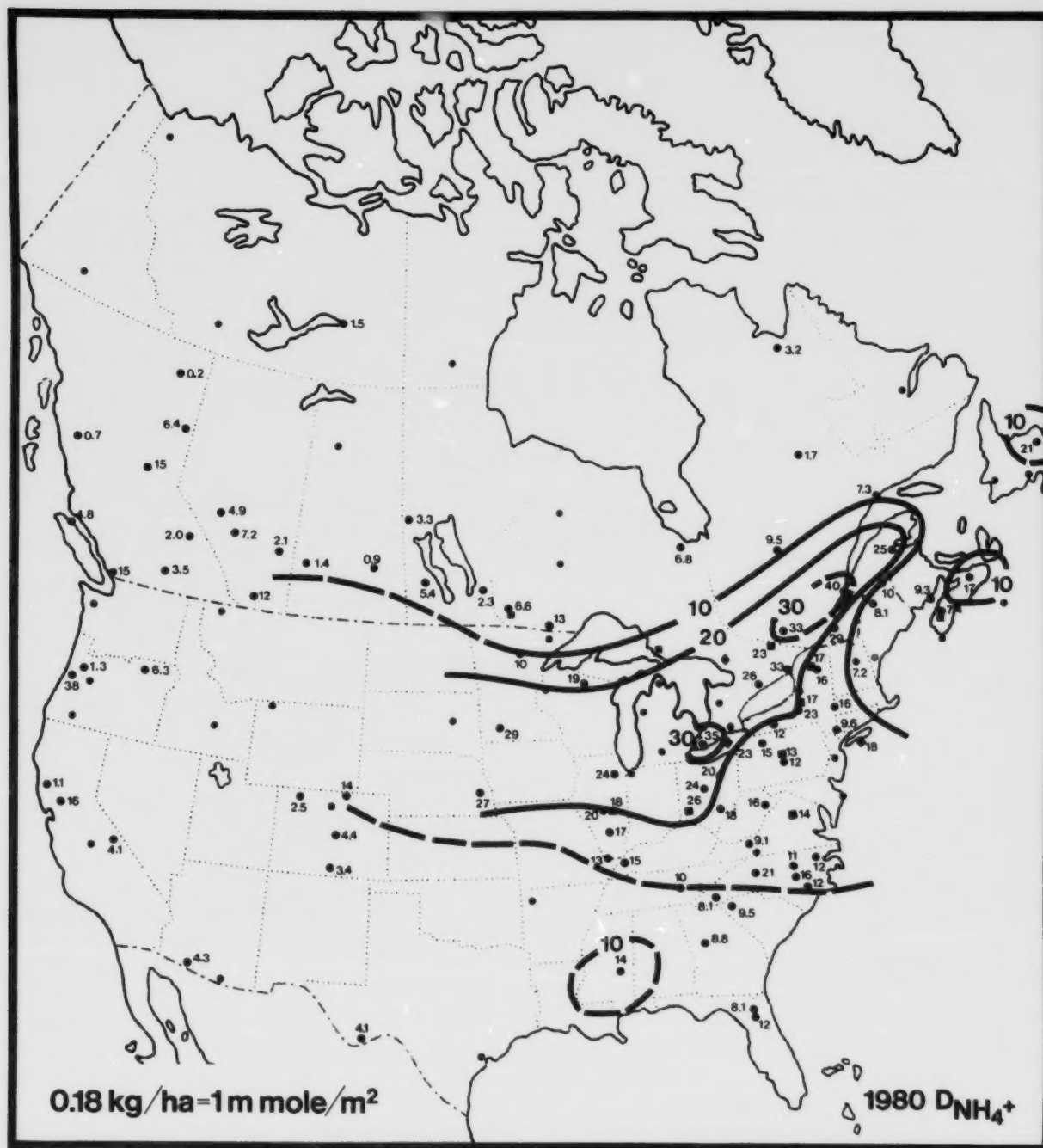


Figure 3.10: The spatial distribution of annual wet deposition of ammonium ion ($\text{mmole} \cdot \text{m}^{-2}$) in North America in 1980. Data from 4 networks are plotted. In Canada: CANSAP (circles) and Air Pollution Monitoring Network (squares). In the United States: National Atmospheric Deposition Program (circles) and MAP3S (squares) (Barrie and Hales 1984).

Alberta ranged from $15.5 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ at Calgary to $6.8 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{y}^{-1}$ at Beaverlodge. Subsequent analysis by Bertram et al. (1986) yields 5 year sulphate deposition averages, over the period from 1980 to 1984, of $9.1 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$. At Calgary, the average sulphate deposition was $11.9 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$, and at Beaverlodge it was $5.8 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$.

Analysis of CANSAP data for the 1977 to 1980 period (Barrie and Sirois 1982) addressed the usefulness and limitations of CANSAP data as well as the geographic and temporal patterns in the data. Among several recommendations in that report was the upgrading and improvement of network operations. Subsequently the CAPMoN network was implemented.

The analysis of CANSAP data by Barrie and Sirois (1982) pointed out the unrealistically high (by a factor of at least 2) calcium and magnesium concentrations in regions having either calcareous soil, or roads salted and sanded in winter, or both. The effect of contamination by calcium and magnesium was greatest in the southern prairies, central Alberta and northeastern British Columbia. In these regions, data for the composition of precipitation for at least the years analyzed 1977 to 1980 were deemed unreliable.

Analysis of the CANSAP data for the southern Prairies for 1977 to 1980 showed that the seasonal variations of all ions except H^+ and NH_4^+ were similar (Barrie and Sirois 1982). Highest levels occurred in winter (November to March) and lowest in summer (July, August). Calcium had a secondary peak in May and June. The H^+ concentrations in this region were low. The influence of dust derived from soil (due to

poor siting and natural processes) requires that extreme caution be used in interpreting the precipitation data for the southern Prairie CANSAP sites. Data for the stations in central Alberta showed no seasonal patterns and similar conditions due to unreliable data apply. In the northeastern British Columbia region, ammonium, calcium, magnesium, sodium, and chloride levels were higher earlier in the year (January to May) while sulphate and nitrate levels varied little throughout the year. The high calcium levels suggest unreliable siting.

The Pacific coast receives precipitation with the highest acidity in western Canada. The levels of all ions except magnesium were highest in the summer-half of the years. The levels of sea salt-derived ions sodium and magnesium were lowest in the least stormy months (June to September). The precipitation data for this region required corrections for sea salt.

In the Canadian Shield region (Northern Saskatchewan and Northwestern Manitoba as defined by Barrie and Sirois 1982), the regional calcium concentrations were low and had little seasonal variation.

3.4.2 Model Estimates of Wet and Dry Deposition in Western Canada

Total deposition of sulphate is an important variable for which there is no direct measurement. Total deposition consists of sulphate in two forms--"wet" and "dry". Measurement of wet deposition is relatively straightforward, but measurement of dry deposition is difficult. Because of

this difficulty, estimates of dry deposition are made by the use of atmospheric models.

There have been three notable attempts to estimate the amount of dry deposition in western Canada. One of the current estimates is based on the AES-LRTAP model, using data for sulphate emissions and wet sulphate deposition in the region in 1978 (Kociuba et al. 1986). The findings of this modelling activity are that: 1. wet deposition predominates in areas far from sulphur sources (wet:dry, 3:1); 2. dry deposition predominates close to sulphur sources (wet:dry, 1:3); and, 3. at intermediate ranges--200 km to 500 km--the two forms of deposition are roughly equivalent proportions of the total (wet:dry, 1:1).

These findings are corroborated by modelling work on the sulphur budget of western Canada (Summers 1986). This work documents the sources of sulphur in the atmosphere, subtracts from this the measurable wet deposition, and assumes that the balance remaining is dry deposition, expressed in units of $g \times 10^{10} S \cdot yr^{-1}$ (Figure 3.11). Note that the geographical positioning of the internal fluxes has no implied significance.

In this model, input terms include sources of sulphur in the atmosphere and background wet deposition. These are: manmade emissions, $E=54$; natural emissions in the form of wind-blown dust, $N=6$; inflow from British Columbia and the U.S.A, $I=8$; and background wet deposition, $B \sim 3$.

Output terms in this model include losses of sulphur from the atmosphere: wet deposition, $W=48$; dry deposition,

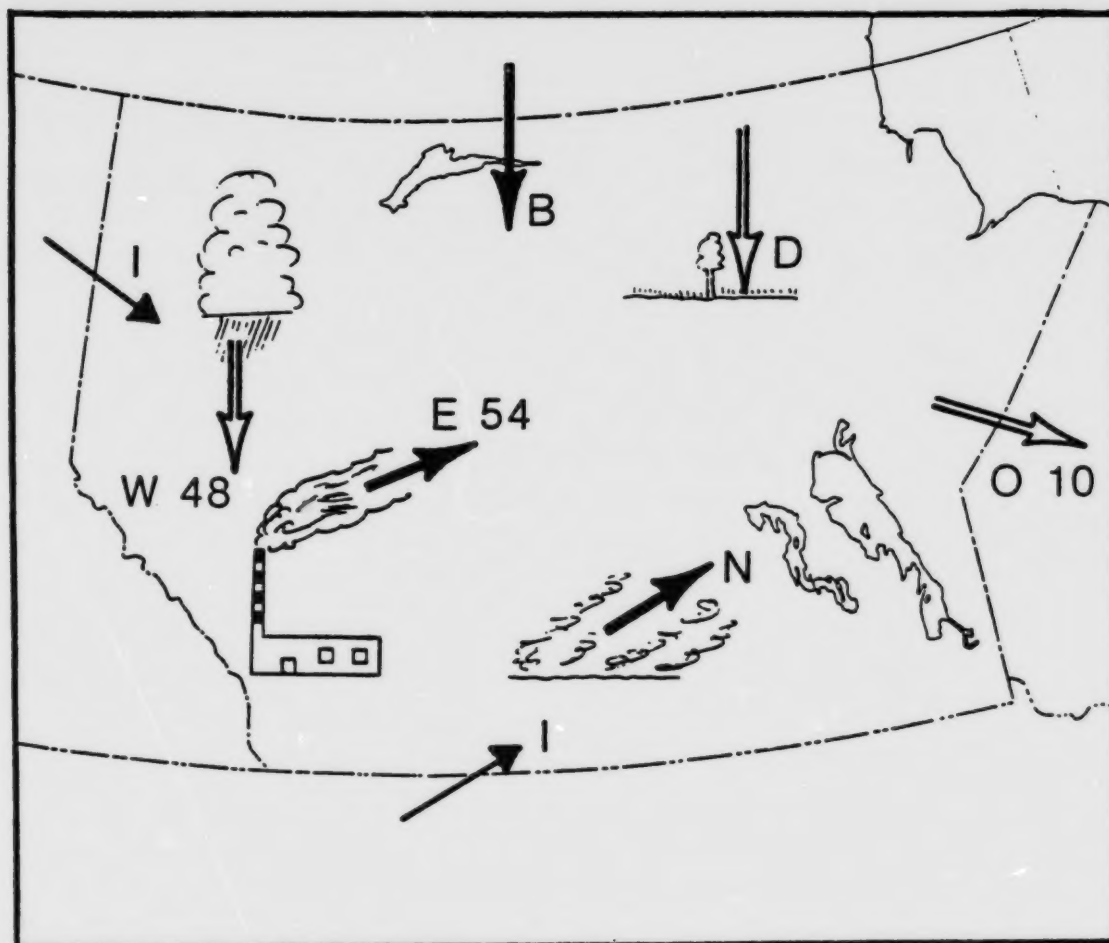


Figure 3.11: Schematic representation* of the major fluxes in the annual sulphur budget for the combined Prairie provinces. Units are $\text{g} \times 10^{10} \text{ S} \cdot \text{yr}^{-1}$. Solid arrows indicate inputs to the atmosphere, open arrows indicate outputs. Previously estimated fluxes are: emissions $E = 54$; wet deposition $W = 48$; outflow to eastern Canada $O = 10$. Remaining terms are: inflow from B.C. and the U.S.A. (I), background wet deposition (B), natural emissions mainly in the form of wind blown dust (N), and dry deposition (D).

*Note that the geographical positioning of the internal fluxes has no implied significance (from Summers 1986).

D=unknown; and outflow to eastern Canada, $O=10$. Dry deposition (D), the unknown quantity in this input/output equation, is: $(54+8+6+3)-(48)=23 \text{ g} \times 10^{10} \text{ S}\cdot\text{yr}^{-1}$ (Summers, pers. comm., Atmospheric Environment Service, Downsview, Ontario).

Based on this estimate, the average ratio of wet deposition to dry deposition, in western Canada, is: (wet:-dry, 48:23), or $\sim 2:1$. Thus the estimate of the ratio of the two forms of sulphate deposition in western Canada made by Summers (1986), agrees well with the estimates made by Kociuba et al. (1986).

A third modelling exercise, involving the same data, highlights the importance of dry deposition near the major ion sources, particularly in Alberta (Sandhu et al. 1986). In these settings, the ratio of wet deposition to dry is $\sim 1:2$ (w:d, 1:1.86 Sandhu 1986). This estimate is consistent with the findings of the other two authors, Summers (1986), and Kociuba (1986).

3.5 Special Studies

The air quality and precipitation data from the federal and provincial networks are complemented by special studies mounted by government or industry or both. The Alberta Government/Industry sponsored "Acid Deposition Research Program" (ADRP) is the most extensive undertaking. This project is directed at providing "a comprehensive understanding of the effects and consequences of acid-forming gases in the environment" in Alberta. The study includes inter alia the acquisition of air quality data at a background and a source-influenced site in Alberta, the pre-

paration of emissions inventories for SO_x and NO_x and the examination of dose response relationships between vegetation and acid-forming pollutants (ADRP 1985). Published data for the study are not yet available, but it is anticipated that the study will provide reliable and comprehensive data on the effects of acid-forming pollutants in Alberta and western Canada, especially Prairie environments.

3.6 Summary

Regional scale modelling and monitoring activities are needed to provide estimates on the loadings of acid-forming pollutants in western Canada. Modelling studies are required to provide estimates for future emission scenarios and to provide accurate regional estimates to complement monitoring data.

Appropriate models have been identified and are under final stages of development. The necessary emissions, meteorological, physical, and chemical input data for the models are generally available, but considerable effort will be required to prepare data in the appropriate formats needed by Lagrangian and Eulerian models. Consistent (same base year) emissions data will be available in the near future.

The existing networks for wet and dry deposition in western Canada are operated by a variety of agencies or institutions (federal, provincial, municipal, industrial). In the case of ambient concentration data from which estimates of dry deposition loadings may be derived, there is inadequate spatial coverage in western Canada. Most existing air monitoring stations are located in urban areas or are sited to provide source-specific data for regulatory purposes

rather than for providing regionally representative measurements.

The existing wet precipitation networks do provide reasonable spatial coverage, except for southern Saskatchewan, and northern British Columbia and in the Northwest Territories.

The networks use a variety of sampling equipment, have different sampling periods ranging from daily to monthly, and appear to have varying quality assurance and quality control protocols. These factors are not likely to provide consistent, reliable, high quality data.

Existing loadings of acid-forming pollutants in western Canada are generally low. Of these pollutants, hydrogen ion, sulphate, and nitrate have similar spatial distribution patterns. However, spatial patterns for ammonium are different. In the southern prairie regions, precipitation chemistry is influenced by alkaline particulate matter in the atmosphere. There, high concentrations of calcium and magnesium result in high pH levels in precipitation. The lowest pH levels are found in coastal British Columbia where there are few alkaline materials in the atmosphere.

Perhaps the most important implication of the data summarized in this chapter is the apparent poor correlation between wet sulphate and precipitation acidity in some regions of western Canada. This means that it is unlikely that a target loading based on wet sulphate (or wet and dry for that matter) can be developed by simple analogy to the eastern Canadian situation.

4.0 CURRENT CLASSIFICATION OF RECEPTOR SENSITIVITY

4.1 Regional Distribution of Ecosystems

The western Canadian provinces contain a variety of ecosystems including agroecosystems, forests, wetlands, and aquatic ecosystems. Because prevailing wind patterns during the growing season are basically from west to east, originating in the north Pacific, each of these ecosystem types can potentially be exposed to sulphur emissions generated in the Pacific Northwest and western Canada (see Figure 3.2).

The region with the greatest agricultural potential in western Canada is located in a crescent-shaped area from Winnipeg to Edmonton and south to Lethbridge (Simpson-Lewis et al. 1979). Prime agricultural land in western Canada is among the most productive in the nation for livestock and grains. The acreage and production of field crops, horticulture, and vegetables in the four western provinces are shown in Table 4.1. Wheat accounts for approximately 60% of the cultivated area in the prairies. In addition to other cereals, including barley (~20%), oats (~5%), and rye (~1%), grains constitute about 85% of the agricultural area east of the Rocky Mountains. Approximately 10% of the area is used for animal feed-grasses and the rest (~5%) is utilized for cultivation of sunflower seeds, potatoes, and other field crops.

Agricultural production in British Columbia is much lower than in other western provinces. Grains and feed-grass account for approximately 70 percent of the production. The

remaining 30 percent of the province's production consists of apples and other fruits (~18%), most of which is grown in the Okanagan Valley, and vegetables (~5%) which are grown throughout the province, and other crops (~7%).

North of this large agricultural area in western Canada are several forested ecozones. The boreal plains, containing mostly aspen parklands and boreal forests, have unlimited capabilities for silviculture, according to the Canada Land Inventory for forests (Simpson-Lewis et al. 1979). By comparison, British Columbia has a complex variety of forested zones, many of which are also very important for forestry. Section 5.4 provides more detail on the species found in the western provinces and the territories.

Large areas of western and northern Canada are covered with wetlands, some in the form of shallow "sloughs" (ranging in size from less than one hectare up to several hundred hectares) and others in the form of muskeg that is typical of much of northern Canada. Within western Canada the highest concentration of wetlands occurs in southcentral and northeastern Manitoba, northern Alberta, northeastern British Columbia, and the southwestern part of the Northwest Territories (Energy, Mines and Resources 1974). More detail on the types of wetlands found in western Canada is provided in Section 5.6.

The surface waters of primary interest to this study are found in regions with a relatively low capability to neutralize acidic precipitation. As outlined on maps presented later in this section, these regions include coastal British Columbia, northern Saskatchewan, northwest

Table 4.1

Area and production of field crops, vegetables and horticulture in the four western provinces

Crop	Manitoba		Saskatchewan		Alberta		British Columbia	
	Area ¹	Prod. ²	Area ¹	Prod. ²	Area ¹	Prod. ²	Area ¹	Prod. ²
Wheat ³	1619	3728	7973	17146	2772	6137	61	114
Barley ³	809	2373	1396	3636	2509	6684	91	189
Flaxseed ³	364	437	202	243	61	76	-	-
Rapeseed ³	344	401	567	726	749	930	57	57
Oats ³	243	571	465	1002	567	1403	28	65
Corn ³	112	735	-	-	15	432	12	485
Rye ³	81	183	188	363	121	241	5	7
Peas ³	44	82	20	38	5	15	-	-
Sunflower seed ³	71	87	6	7	-	-	-	-
Potatoes ³	16	263	1	22	7	192	4	94
Mustard seed ³	10	11	36	42	20	27	-	-
Sugar beets ³	11	372	-	-	12	469	-	-
Apples ⁴	-	-	-	-	-	-	NA	160.5
Other fruits ⁴	-	-	-	-	-	-	NA	70.2
Various veg. ⁴	0.4	11	0.7	7.5	0.8	11.5	4	56

¹ x1000 hectares.

² x1000 hectares.

³ 1982 data.

⁴ 1984 data.

⁵ Data are based on Canada Year Book (1985) and Statistics Canada, Agriculture/Natural Resources Division (1986).

NA Not available.

- No data.

Prod. = production.

veg. = vegetables.

and southeast Manitoba, and much of the eastern part of the Northwest Territories. Within these regions are several important fisheries, most notably salmon on the west coast and lake whitefish, walleye, sauger, and lake trout in inland areas. Other uses of surface water potentially affected by acidic precipitation include a wide diversity of water-dependent wildlife, drinking water and irrigation supplies, and water-based recreation.

Within each major ecosystem type (listed above) are a number of components, including, for example, natural vegetation, agricultural crops, as well as surface and ground waters, that may differ greatly in their sensitivity to atmospheric inputs. For example, air pollutants can affect vegetative components either directly via impingement or indirectly through interactions with soils and runoff. In some instances, indirect effects may be more influential, especially over longer time periods. Therefore, in addition to knowing which types of vegetation are present in a region, and their sensitivity to direct pollutant effects, it is also essential to know how the soils of these regions will respond to the deposition of atmospheric pollutants. Similarly, the hydrological flowpath of water through soil, and the chemical response of soils (on both short and long time scales), are critical in determining aquatic ecosystem response.

Available data on soil characteristics and chemistry are best for areas currently used for agriculture. Unfortunately, detailed soils (and to a lesser extent geological) data for more remote and northern, "sensitive" areas are often lacking even though they may be very important because some of the largest regional point sources are in such areas. Nevertheless, enough is known about the

western Canadian provinces and northern territories to at least create regional-level soils and geology maps.

In western and northern Canada, selected representative ecosystems have been designated as national or provincial parks. Currently, there are fifteen national parks and nearly sixty provincial parks in this portion of the country; many of these parks are located in areas considered to be sensitive to acidic deposition. There are, in addition, over 250 (proposed or established) International Biological Program (IBP) ecological sites in this region, and seventy-five or more established ecological reserves in British Columbia (Simpson-Lewis et al. 1979). Thus far, no special attempts have been made to either determine the sensitivity of these protected areas to atmospheric pollutants or to establish long-term monitoring sites that could provide "early warning" signals of deteriorating conditions.

4.2 Mapping Receptor Sensitivity

Receptors refer to each of the biotic and abiotic components of terrestrial and aquatic ecosystems; generally, these are bedrock, soils, vegetation, and water. Although common to all ecosystems, these components exist in different proportions, particular to each ecosystem. Thus, the distribution of receptors is physiographically distinct.

4.2.1 Geochemistry: Bedrock and Soils

Unlike the situation in eastern Canada, soils and mapping in western Canada have tended to be based on a definition of sensitivity that emphasizes the capacity of

soils to resist change in pH and base saturation (Wiens 1983), both factors having the capacity to influence plant growth. At the same time, these efforts have been complemented by the application of techniques developed in eastern Canada for mapping the capacity of soils and geological formations to modify the input of atmospherically deposited acids to aquatic ecosystems. This has resulted in the independent production of two sets of maps, both of which are presented below. Readers are cautioned that these are preliminary (map borders are subject to change), with final maps to be available in 1987.

Soil and Bedrock Potential to Reduce Acidity

Silicate minerals neutralize acids through hydrolytic weathering reactions that occur at different rates depending upon the intrinsic properties of each specific mineral. As well, the pool of base cations accumulated in soils and the physio-chemical characteristics of the soil matrix are also key factors in determining the capacity of the terrestrial substrate to neutralize incoming acidity. Of the four major processes controlling surface water acidification, identified by the U.S. National Research Council Panel on Process of Lake Acidification (NRC 1984), two involve reactions between soil water and soils. These are:

- 1) release of base cations to water flowing through soils; and
- 2) retention of sulphate (and other anions) by the watershed.

Using soils and geology information it is, therefore, possible to construct maps depicting regional patterns

reflecting "susceptibility" of aquatic systems to changes caused by acidic deposition. This has been done for western Canada (Wiens et al. 1987) with the results presented in Figure 4.1. The soil and bedrock factors shown in Table 4.2 served as the basis for defining the ecosystem map units regarding their low, medium, or high ability to reduce acidity. The derived criteria are defined in Wiens (1983) and are premised on the guidelines discussed in MOI (1983). Two additional categories were also used in the mapping efforts--peatlands and ice-covered areas--to reflect the special nature of these landforms vis a vis acidic precipitation.

Examination of Figure 4.1 reveals that many lands in the Yukon, the eastern portion of the Northwest Territories, the northern half of Saskatchewan, northwestern and southeastern Manitoba, and coastal British Columbia have a low potential to ameliorate incoming atmospheric acidity. In comparison, lands in the southern half of Saskatchewan, most of Alberta, much of northeastern and southwestern Manitoba, northeastern and central British Columbia, and the western portion of the Northwest Territories have a moderate to high capacity to neutralize incoming acidity. Areas marked as peatlands and icefields have not been explicitly classified as to their ability to reduce acidic deposition, although wetland sensitivity is discussed further in Section 5.6.

Inconsistencies in mapping, seen most clearly at provincial boundaries, reflect both (1) the fact that despite adoption of standardized protocols, slight differences in their interpretation have resulted in maps that appear to contain errors; and (2) the coarse scale of these small maps. These inconsistencies reduce the reliability of conclusions drawn from comparisons among maps.

Refer to the Map Pocket at the End of this Report

Figure 4.1: Map of the potential of soils and bedrock to reduce the acidity of atmospheric deposition in western Canada.

Table 4.2

Factors used to interpret the potential of terrestrial systems to reduce atmospherically deposited acidity
(After Lucas and Cowell 1984; Shilts 1981)

Terrestrial Factors/Surrogates	Categories
<u>Soil Chemistry</u> Surrogates: Texture Carbonate content Glacial landforms Organic soils	sand, loam, clay high, low, absent >50% of mapping unit
<u>Soil Depth</u>	shallow (25cm - 1m) deep (<1m)
<u>Bedrock Geology</u> Lithology	<u>Type 1:</u> limestone, marble, dolomite <u>Type 2:</u> shale, limestone, siltstone, dolomite, quartzose sandstone with carbonates <u>Type 3:</u> black shale, slate, chert, gabbro, diorite mafic volcanics <u>Type 4:</u> granite, gneiss, quartzose sandstone sydnitic and associated alkalic rocks
% Exposed	<25cm deep

Soil Sensitivity to Acidic Deposition

Interpretation of soil sensitivity to acidic inputs was based mainly on two soil properties--cation exchange capacity (CEC) and pH. Available data on these two properties were coupled with understanding of basic soil chemistry principles and additional information on the soils resulting in subjective ratings of the sensitivity that mineral soils would show to base loss, acidification, and aluminum solubilization. This resulted in an estimation of overall sensitivity (Fessenden 1985) as outlined in Table 4.3. To generate a regional map covering both mineral and organic soils (Figure 4.2), separate criteria (comparable to those shown in Table 4.3) were also used for organic soils (N. Holowaychuk, Alberta Research Council, pers. comm.). When dealing with these organic soils, it also was necessary to compensate for the low bulk density to prevent uncomparably high CEC values since the values are expressed on a per weight basis.

The regional map (Figure 4.2) was prepared in two steps. First, each of the provinces and territories prepared their own soil sensitivity maps. These maps were then combined to create Figure 4.2. The boundary discontinuities seen in this map are due, in part, to the fact that when the organic soils were mapped, somewhat different criteria were used for the organic soils in the preparation of the individual maps (N. Holowaychuk, Alberta Research Council, pers. comm.).

High sensitivity soils are generally very acidic (pH <4.6) and, in the generated map, this category includes Sphagnum-dominated organic soils. Mineral soils with a low

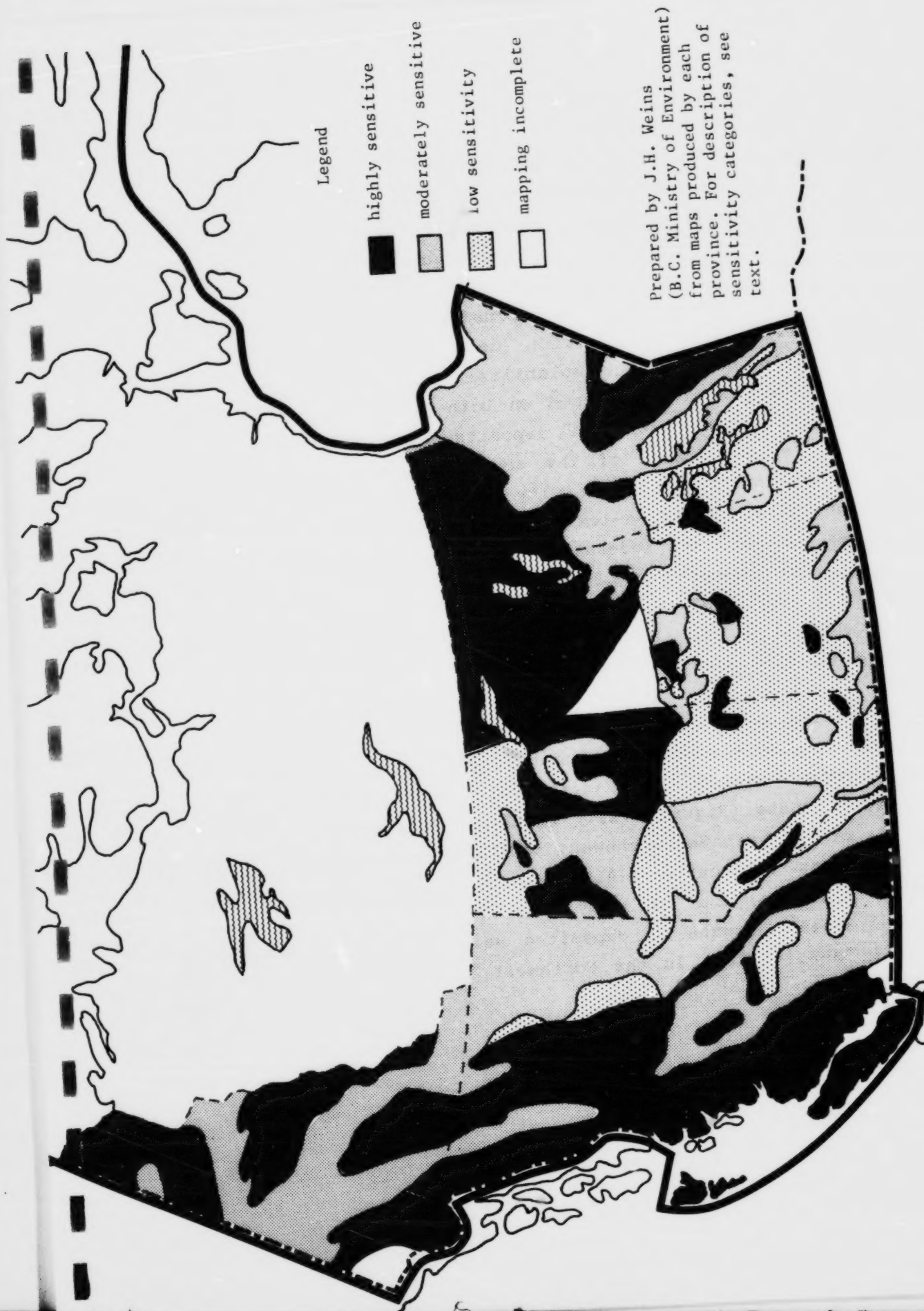
Table 4.3

Criteria for rating the sensitivity of mineral soils to acidic deposition

CEC ¹ (meq•100g ⁻¹)	pH	Sensitivity to			Overall Sensitivity
		Base Loss	Acidification	Aluminum Solubilization	
<6	<4.6	H	L	H	H
	4.6-5.0	H	L	H	H
	5.1-5.5	H	M	H	H
	5.6-6.0	H	H	M	H
	6.1-6.5	H	H	L	H
	>6.5	L	L	L	L
6-15	>4.6	H	L	H	H
	4.6-5.0	M	L	H	M
	5.1-5.5	M	L-M	M	M
	5.6-6.0	M	L-M	L-M	M
	>6.0	L	L	L	L
>15	>4.6	H	L	H	H
	4.6-5.0	M	L	H	M
	5.1-5.5	M	L	M	M
	5.6-6.0	L	L-M	L-M	L
	>6.0	L	L	L	L

Source: Fessenden 1985.

¹ CEC = cation exchange capacity.



Prepared by J.H. Weins
 (B.C. Ministry of Environment)
 from maps produced by each
 province. For description of
 sensitivity categories, see
 text.

Figure 4.2: Preliminary map of soil sensitivity to acidic deposition.

cation exchange capacity (CEC) ($<6 \text{ meq} \cdot 100 \text{ g}^{-1}$) with pH less than 6.6 are also rated as being highly sensitive. Examples of the latter include most coarse-textured soils in the boreal forest region, and to some extent elsewhere. Soils in the moderately sensitive category frequently have moderate to high CEC ($6-15 \text{ meq} \cdot 100 \text{ g}^{-1}$) and are strongly acidic (pH 4.6-5.5). Such soils are considered to be moderately susceptible to base cation depletion but are highly susceptible to aluminum solubilization. Included in this category are soils developed on both glaciolacustrine clays and low-lime content till deposits throughout the boreal forest region. Most of the southern prairie agricultural soils are in the low sensitivity category. These are usually medium textured to fine-textured, have medium to high organic matter content, are moderately acidic to alkaline (pH 5.6 to >6.0), and generally have high cation exchange capacity ($\text{CEC} >15 \text{ meq} \cdot 100 \text{ g}^{-1}$). These soils are rated as having low susceptibility to acidification, base cation loss, and aluminum mobilization.

Soils covering much of British Columbia, the Yukon, northern Saskatchewan, northcentral Alberta, as well as parts of northern and eastern Manitoba are considered sensitive to acidic inputs (Figure 4.2). Equally important, the soils of most of southern Saskatchewan, southern and western Manitoba, and parts of southern Alberta are highly insensitive to acidic inputs; that is, they have the capacity to neutralize substantial amounts of deposited acids. Data to determine soil "sensitivity" in the Northwest Territories are not yet available.

4.2.2 Forests

The lands of western and northern Canada encompass a wide range of ecozones; thirteen of the fifteen ecozones recognized by Environment Canada (Bird and Rapport 1986) are found in this area (Figure 4.3). Of these zones, five are important for forestry: boreal cordillera, Pacific maritime, montane cordillera, boreal plain, and the boreal shield. Current scientific understanding of the degree to which terrestrial biota are susceptible to acidic precipitation is still fragmentary, making it difficult to apply information collected elsewhere when dealing with areas where little is known of how individual species will respond to acidification-related stressors. For this reason (and others), no attempts have yet been made to classify and map the sensitivity of western Canadian forests to atmospheric pollutants.

4.2.3 Surface Waters

Figures 4.4 and 4.5 are preliminary maps of the sensitivity of lakes in western Canada, based on lake calcium and alkalinity concentrations, respectively. [Alkalinity is operationally defined as the sum (in microequivalents) of all the bases that can be titrated with strong acid to a preselected equivalence point or reference level (Driscoll and Bisogni 1984)]. Of the two lake sensitivity maps, the alkalinity map (Figure 4.5) is more directly relevant to defining aquatic sensitivity, since lakes with zero alkalinity can still have substantial calcium.

The preliminary nature of these maps, however, should be stressed. The criteria for sensitivity ranking were

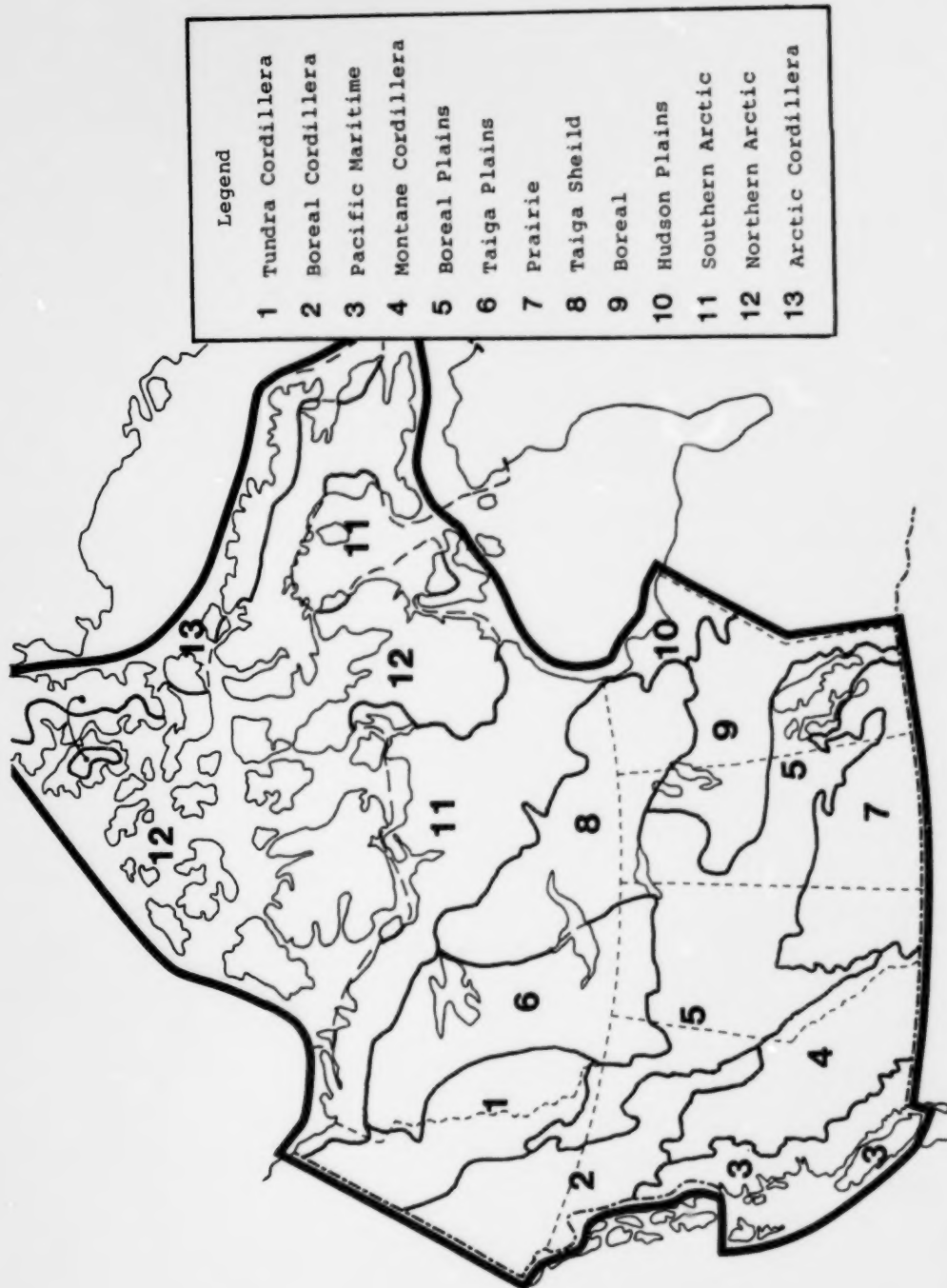


Figure 4.3: Terrestrial ecozones of western Canada. Source: Bird and Rapport 1986.

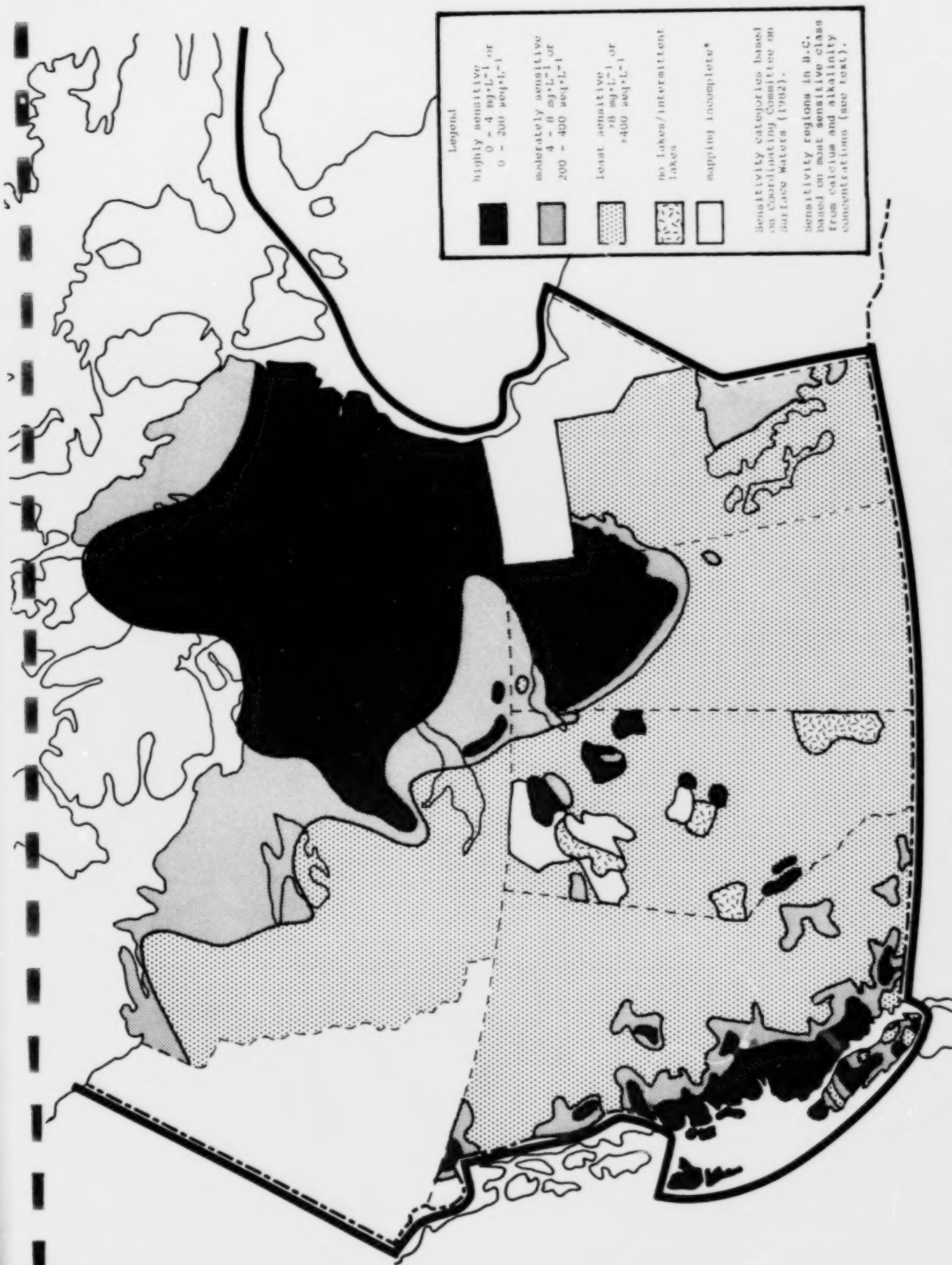


Figure 4.4. Preliminary map of lake sensitivity to acidification, based on lake calcium concentrations.

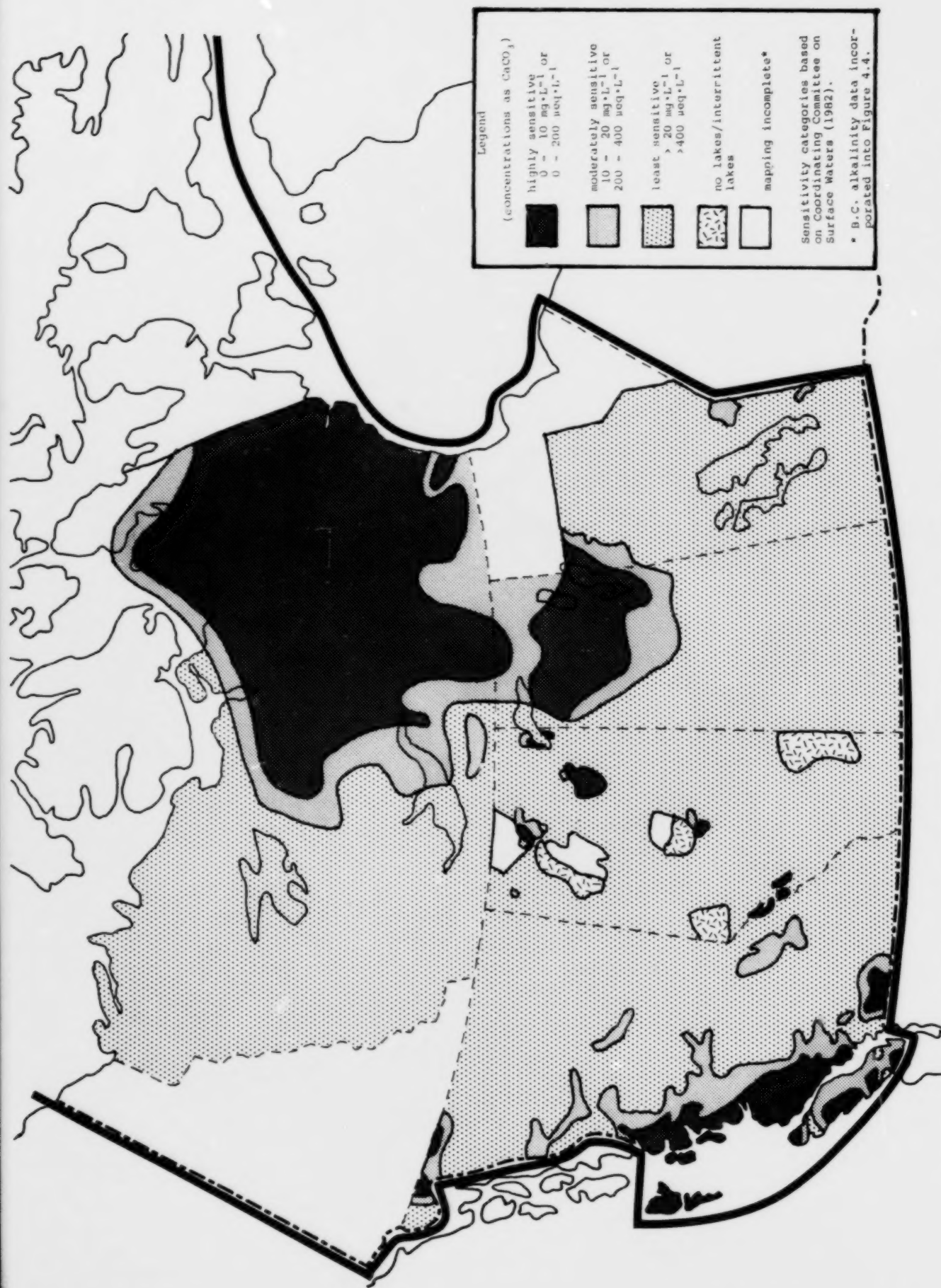


Figure 4.5. Preliminary map of lake sensitivity to acidification, based on lake alkalinity.

developed by the Coordinating Committee on Surface Waters (1982), based on Harvey et al. (1981). A substantial number of gaps in the geographic coverage of water quality data were identified by the Coordinating Committee in their 1982 report. Data have now been collected to fill most of these gaps (Technical Committee for LRTAP in Western Canada 1984), but maps for the Yukon and parts of northern Alberta were not available in time for this report.

Although within the last three years the provincial agencies have greatly improved both the quality and geographic coverage of their surface water databases, there has not yet been an assessment of the intercomparability of these data and the relative levels of uncertainty associated with different regions' classifications. Also, the sensitivity classes used in Figures 4.4 and 4.5 are quite broad, and do not permit evaluation of the abundance of lakes at greatest risk (i.e., those lakes with less than 50 $\mu\text{eq}\cdot\text{L}^{-1}$ alkalinity, and those with 50-100 $\mu\text{eq}\cdot\text{L}^{-1}$).

In some regions of western Canada it is possible to verify the estimates of resources at risk. How well this can be done depends on the quality of the data in existing water quality databases, and on the ease with which bias in the databases can be detected. Historical databases are inevitably biased; but the nature of the bias is not necessarily disclosed in the published documentation. This is the case with Alberta and Manitoba.

The water quality databases of British Columbia and Saskatchewan have biases that are known; but these provinces have taken some corrective action. For example, the British

Columbia database contained data that had been collected largely along established roads, south of the 56th parallel. Thus, there were few data in remote places--off established roads, and north of the 56th parallel. Sampling programs initiated by the Province in the early 1980's have begun to correct these deficiencies. Where the known biases have been rectified, estimates of the resources at risk can be made with greater assurance.

In spite of the preliminary nature of the aquatic sensitivity maps, the areas considered to be highly sensitive in Figure 4.4 (Queen Charlotte Islands, coastal British Columbia and Vancouver Island, pockets in northeastern Alberta, northern Saskatchewan, the eastern part of the Northwest Territories) significantly overlap with the regions ranked as having a low potential for soils and bedrock to neutralize acidity (Figure 4.1). This congruence lends some confidence to both the soils and bedrock map and the lake sensitivity map. There are, however, some differences; the soils and bedrock map generally indicates a greater region of sensitive terrain:

- 1) in southeastern British Columbia, only a very small area is classified as being in the highly sensitive category in the lake sensitivity map (Figure 4.4); on the soils and bedrock map a substantial area is considered to have a low potential for neutralizing acidity;
- 2) the area of northern Saskatchewan ranked as being highly sensitive in Figures 4.4 and 4.5 is smaller than the area with low neutralization potential in Figure 4.1; and

- 3) the area of highly sensitive aquatic systems in the Northwest Territories in Figure 4.4 is smaller than the low neutralization zone in Figure 4.1.

Differences in the two maps are due largely to the process of breaking up continuous distributions of sensitivity into discrete categories. Many of the regions noted above were ranked moderately sensitive rather than highly sensitive on the lake sensitivity map. Other factors which might also contribute to the lack of correspondence include:

- climatic factors modifying the effects of soil and geology (e.g., high dilution volumes);
- spatial heterogeneity in soil and geology, which affect lake chemistry but do not show up on generalized soil and geology sensitivity maps;
- relatively low deposition throughout much of western Canada, so that lakes within areas with soils and geology of low neutralization capability still retain moderate alkalinity levels; and
- generalizations inherent in the mapping process.

The existing maps provide a useful guide to help focus attention on those regions potentially most vulnerable to acidic deposition, from a scientific perspective. The actual and potential economic worth of various regions, however, may also provide another basis for deciding which geographic areas to examine. Assuming the importance of quantifying the ecological resource at risk, it would be necessary to:

- 1) use sampling of topographical maps (as in Minns 1984) to estimate the size of the surface water resource (number and area of lakes, kilometers of streams) within each of the zones considered to be highly sensitive;
- 2) examine existing datasets within these zones to assess how representative they are (based on location, lake area, alkalinity, and soil/bedrock type);
- 3) fill data gaps and correct biases by stratified random sampling of the existing datasets (that is, a simple random sample from each dataset, supplementary sampling, or entirely new surveys;
- 4) produce frequency distributions of lake alkalinity for each of the zones considered to be highly sensitive; and
- 5) develop finer scale maps of sensitivity (e.g., 0-50, 50-100, 100-200 $\mu\text{eq}\cdot\text{L}^{-1}$) for the highly sensitive zones, using a consistent and objective mapping procedure which explicitly shows the density of sampling.

In addition, it is important to recognize that although the focus to date has been on lake chemistry, these data do not adequately represent the chemical conditions in (and hence susceptibility of) streams. Running waters can be especially sensitive to rapid acidification during springmelt and rainfall episodes (Marmorek et al. 1986). Due to the short-term nature of these episodes, surveys to assess stream

sensitivity require greater sampling intensity than lake surveys. If it is not feasible to conduct extensive stream surveys, then the preferred approach is to intensively monitor a few calibrated watersheds in regions classified as being highly sensitive.

A critical issue is how to handle data gaps and potential biases in existing data. In his review of the draft version of this report, Dr. Arne Henriksen commented:

"Instead of reviewing old data bases for quality and filling in possible gaps in data sets, I suggest that a regional lake (and stream) survey is carried out in order to establish the present water quality situation, and to obtain a database to which existing steady-state models could be applied."

Though some regions of western Canada appear to have recent surveys that are adequate (e.g., much of British Columbia), other regions may not. For example, the water quality data used for mapping in Manitoba was collected mostly from fisheries and eutrophication studies, both of which tend to be biased toward higher alkalinity lakes.

Fortunately, much is now known about the proper design of surveys to assess aquatic sensitivity, from studies completed in Scandinavia (Overrein et al. 1980), and in North America (including eastern and western Canada) so, the issue of data gaps and other inadequacies is not likely to have serious long-term consequences.

4.3 Summary

The maps presented in this chapter are the best currently available and represent a considerable step forward in our ability to visualize the spatial distribution and extent of western Canadian and northern resources potentially at risk from acidic precipitation. Perhaps one of the most important uses of these maps is to help in deciding where to concentrate future monitoring and evaluation efforts.

5.0 REVIEW OF DOSE-RESPONSE RELATIONSHIPS FOR WESTERN CANADA

5.1 Introduction

Acidic deposition can have both direct and indirect effects on various ecosystem components. For example, sulphur compounds deposited directly onto vegetation or into lakes (with large surface area to watershed ratios) can cause changes in the function or structure of these ecosystems. In such cases, geologic materials (rocks, soils) in the landscape have relatively little influence on the impact processes. In other cases, acidic solutions and materials deposited on the earth's surface can be at least partially neutralized through contact with rocks, soils, and even biologic materials. Thus, effects of acidic deposition on many surface waters and some terrestrial ecosystems can be modulated by the nature and amounts of minerals found in the watershed.

Evaluating the situation in western Canada involves an examination of:

- 1) deposition patterns;
- 2) the sensitivity of geologic and soil materials; and
- 3) the sensitivity of vegetation and surface waters.

These factors and their interactions are illustrated in Figure 5.1. In areas of western Canada where deposition rates are currently high we are, of course, most immediately concerned with situations represented by portion 7 of Figure

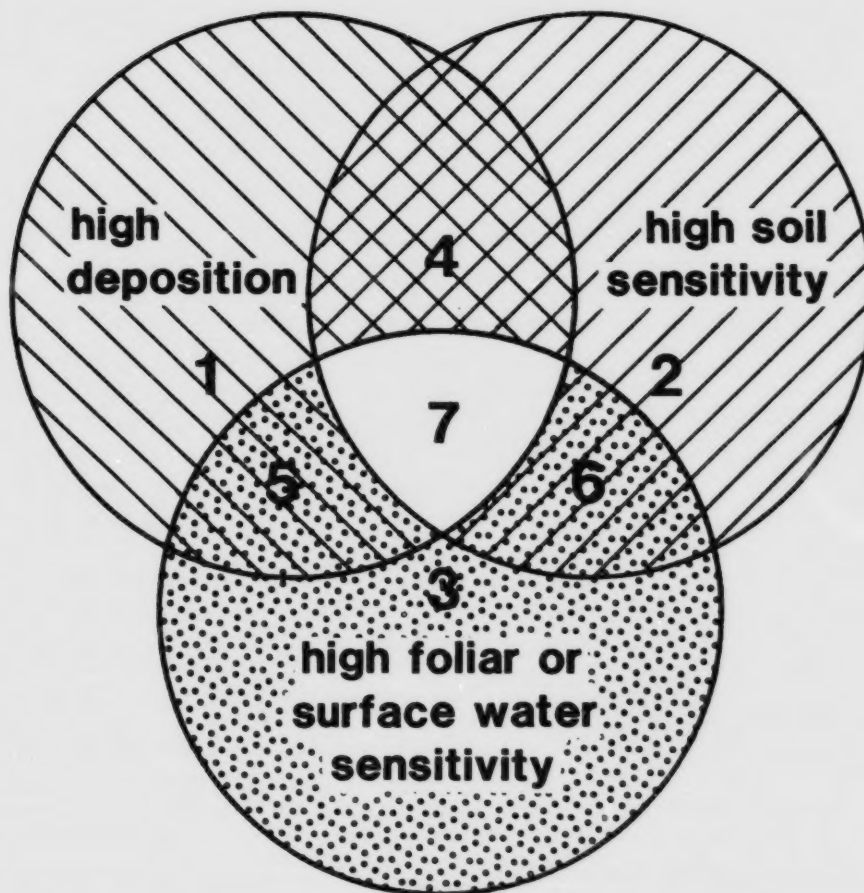


Figure 5.1: Conceptual diagram depicting overlap between areas with high rates of acidic deposition, highly sensitive soils, and containing sensitive freshwater and terrestrial ecosystems.

5.1; fortunately, there are few such areas at the present time. In such regions, those ecosystems represented by portions 4 and 5 also deserve contemporaneous attention. Although much of western Canada is currently not exposed to such high levels of acidic precipitation, deposition patterns are likely to change in the future. Thus, it is important that areas represented by portions 2, 3, and 6 of the diagram be monitored to detect changing conditions. Furthermore, since much of western Canada is currently being exposed to low level inputs of acidic substances, such monitoring is also valuable for helping to document ecosystem responses to low dosages, an integral part of any dose-response relationship.

The remainder of this chapter is organized into five parts, representing the main receptors in western and northern Canada; these receptors are: bedrock and soils, agriculture, forests, aquatic systems, and wetlands.

5.2 Bedrock Geology and Soils

Neutralization of acidic deposition through contact with rocks and soils is believed to be primarily controlled by:

- 1) availability of weatherable minerals and weathering rates;
- 2) base cation buffering and sulphate adsorption capacities of soils;
- 3) depth of soil/till; and

- 4) soil hydrologic characteristics (hydrologic pathways and water flow rates).

Additional factors, such as atmospheric inputs, salt-effect alkalinity reduction (Seip 1980; Reuss and Johnson 1985), type of forest cover, and in-stream and in-lake processes, also help control soil and water acidification (Church and Turner 1986). Two key questions relating to the neutralization of acidic deposition by geologic materials are:

- 1) At what rate do soils and geologic materials neutralize acidic deposition?
- 2) How long can these neutralization reactions be sustained?

5.2.1 Bedrock Geology

From a geological perspective, a key question relating to the dose-response behaviour of bedrock exposed to acidic deposition is the effect on weathering rates of varying the acid dose. This is because, in the long run, chemical weathering of bedrock minerals is the rate-limiting step in the supply of basic cations for export from watersheds (Schnoor and Stumm 1985). Unfortunately, as in most other regions, there are few data pertaining to weathering rates of rocks found in western Canada. Consequently, it is not possible to construct dose-response curves relating hydrogen ion activity in soil solutions to chemical weathering rates for most of the rock types found in this region. Nevertheless, there are some data for selected locations. For

example, silicate rocks in the Jamieson Creek area of British Columbia are currently hydrolyzing at the rate of $0.3 \mu\text{eq}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ (Zeman 1973). This is considerably slower than rates observed at South Cascade Glacier in nearby Washington State. At this latter site, silicate hydrolysis is occurring at a rate of $0.9 \mu\text{eq}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ (Reynolds and Johnson 1972). The difference in weathering rates has been attributed by Johnson (1984) to the fact that fresh, chemically active crustal surfaces are continuously created by glacial action at the South Cascade Glacier, whereas rocks at Jamieson Creek are covered with a rime developed over the past 10,000 years. It is assumed that weathering rates for both these sites represent "background" values; what effect increasing atmospheric deposition of acids would have on these rates is unknown. The reader is also cautioned that these weathering rates may be atypical of much of western Canada, especially those siliceous areas containing lakes with alkalinity $<100 \mu\text{eq}\cdot\text{L}^{-1}$. Furthermore, depending upon the use to which the weathering rate information will be put, these data may be required for depths of up to several meters or for just the rooting zone.

5.2.2 Soils

Due to their inherent buffering capacity, soils can be expected to be less responsive to short-term acidic precipitation events than either exposed vegetation or surface waters, particularly waters subject to episodic acidification (Marmorek et al. 1986). Consequently, effects of acidic deposition on soils are probably best assessed in terms of deposition rates rather than on the basis of precipitation concentration (Evans et al. 1981).

Evans et al. (1981) identified two main areas of concern with respect to effects of acidic deposition on soils: (1) effects on nutrient leaching and soil acidification, with subsequent consequences for nutrient status of terrestrial systems; and (2) the modifying influence of soils on the chemistry of atmospheric precipitation, which ultimately is reflected in the composition of surface waters. These two processes are not wholly independent. For example:

- 1) Cation exchange reactions involving incoming hydrogen ions result in neutralization of acidic inputs, but at the cost of displacement and leaching of nutrient cations.
- 2) In very acidic soils, cation exchange reactions with incoming hydrogen ions may have only a minimal effect on soil nutrient content (Krug and Frink 1983), but may result in mobilization of aluminum ions which can be toxic to root hairs (e.g., Schier 1985) and aquatic biota (Baker 1984).

Sulphur, a major component of acidic deposition, is generally considered to be the most important chemical species responsible for long-term acidification of surface waters (NAS 1984). Sulphur is also a macro nutrient in both agricultural (Terman 1978) and forested ecosystems (Johnson et al. 1980; Turner et al. 1980), with atmospheric inputs serving as an important source of this nutrient. Thus, it is important to recognize the potential paradox between reducing sulphur inputs to protect ecosystems and allowing increased atmospheric deposition of this essential element to achieve a fertilizer effect.

Since the early 1900s, scientists have recognized both the acidifying effect of sulphur dioxide and the nutrient effect of atmospheric sulphur fertilization on soils (Katz et al. 1939). Katz et al. (1939) investigated the chemistry of soils around Trail, British Columbia where over 40 years of smelter operations had resulted in severe damage to the surrounding vegetation. Based on information in Katz's report to the National Research Council of Canada, we see that sulphur loadings at that time were on the order of $9 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$, exclusive of direct sulphur dioxide uptake. Even at that time it was recognized that:

"In cultivated soils the effect of exposure on soil acidity was negligible, but in forest soils...a gradual loss of calcium and other bases was evident if they were exposed to long-continued fumigation with high concentrations of sulphur dioxide." (Katz et al. 1939: p. 132).

Unfortunately, Katz's study suffered from a problem which still persists--extremely few studies examining the effects of sulphur deposition have been able to adequately document the dosage responsible for observed effects. Consequently, although we are able to carefully document the effects of atmospheric deposition of elemental sulphur (e.g., Addison et al. 1984a), ammonium sulphate (van Breeman et al. 1982), and wet and dry sulphate deposition in soils (e.g., Richter et al. 1983; Church and Turner 1986), it is still impossible to clearly establish dose-response relationships between sulphur inputs and response indicators like basic cation loss or aluminum mobilization, even for specific soils.

With the above caution in mind, some major aspects of the sulphur cycle in western Canadian soils are reviewed, vis a vis target loadings. Specifically, three major soil processes are examined:

- 1) sulphate mobility;
- 2) sulphur mineralization; and
- 3) biological uptake.

Factors affecting sulphur cycling within ecosystems, particularly those influencing sulphate mobility, are of primary concern in the study of surface water acidification (NAS 1984). Reuss and Johnson (1985) have discussed the role mobile sulphate ions in runoff play as "carrier anions", enabling the leaching of cations. In a careful and exhaustive review of this concept, Church and Turner (1986) indicate that in areas receiving acidic deposition, sulphate mobility is probably largely responsible for controlling leaching of base cations, aluminum complexes, and hydrogen ions to surface waters. Sulphate mobility is influenced by adsorption and desorptive processes in soils, hydrologic flow path within soils, and soil depth (Church and Turner 1986). While all these factors are undoubtedly important, the database for western Canadian soils is inadequate to assess them individually. An alternative is to examine input-output data for sulphur at selected calibrated watersheds.

Sulphur input-output budgets are available for five forested sites in British Columbia and Alberta (Table 5.1). At all five locations, sulphur output exceeds sulphur input. This indicates that soils in these forested mountain watersheds are not currently acting as sulphur sinks. This is

Table 5.1

Average annual sulphate input-output budgets for sites in western Canada $\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$. (After Rochelle et al. 1986)

Watershed Location	Input	Output	Retention	Atmospheric Contribution (%)
Haney, British Columbia				
WS - A	6.1 (B)	6.7 (S)	- 0.6	91
WS - B	9.7 (B)	12.2 (S)	- 2.5	80
Jamieson Creek, British Columbia	6.7 (B)	9.0 (S)	- 2.3	74
Carnation Creek, British Columbia	8.7 (B)	28.2 (S)	-19.5	31
Marmot Basin, Alberta	6.3 (W)	17.4 (S)	-11.1	36

(B) = bulk precipitation; (W) = wet fall; (S) = streamflow.

WS = watershed (drainage basin).

consistent with the pattern described by Rochelle et al. (1986) where sulphate retention is seldom observed in areas containing relatively young soils, such as the entire region north of the Wisconsin glaciation limit. This suggests that elevated sulphur loadings into western and northern Canadian watersheds with similar soils will result in increased concentration of mobile sulphate ions in soil waters. This, in turn, is likely to lead to leaching of plant nutrients from affected forest soils.

Sulphur input-output budgets may also be available from watershed studies carried out on Baptiste and Wabamun lakes by Saskatchewan's Water Quality Control Branch (1976-1982). Since the data from these studies include export of major ions, in forested and agricultural watersheds, estimates of ion budgets and weathering rates may be possible in both these settings.

Another possible explanation for the observed sulphur export from the calibrated watersheds in western Canada is sulphur mineralization. In studies at Marmot Basin, Alberta, Mitchell et al. (1986) found that atmospheric sulphur inputs were much less important (31%) than sulphur released by the weathering of shales in the watershed (69%). This is probably not uncommon throughout the Rocky Mountains, for Gibson et al. (1983) also found a number of streams in Rocky Mountain National Park, Colorado that had elevated sulphate concentrations despite relatively low atmospheric sulphur inputs. These streams were found to be located in basins containing sulphur-rich minerals (e.g., shales). In contrast, at Jamieson Creek in the Coast Range Mountains (British Columbia), atmospheric sulphur influx accounts for

nearly three-quarters of the sulphate efflux. Again, the remainder is attributed to oxidation of trace sulphide minerals in the bedrock and till of the watershed (Driscoll et al. 1988).

Unlike the eastern forest soils that contribute relatively low amounts of organic acids (Cronan et al. 1978), soils in the Pacific Northwest tend to contribute moderate amounts of naturally occurring organic acids to soil water solutions. This can result in acidic surface waters ($\text{pH} < 5$), especially in low-order streams draining shallow, acidic soils underlain by silicate rocks. Such a situation has been documented by Driscoll et al. (1988) for the headwaters of Jamieson Creek.

In neutral or calcareous prairie soils, no significant quantities of adsorbed sulphate were measured in studies by Bettany et al. (1974). Carbon-bonded sulphur, however, accounts for between 40% and 70% of the total soil sulphur in grasslands (Maynard et al. 1984); the corresponding value is over 60% for some Rocky Mountain soils in Alberta (Mitchell et al. 1986). These organic constituents provide a clue to yet another factor that must be considered in evaluating the fate of atmospheric sulphur inputs. Recent studies (Swank et al. 1984; Mitchell et al. 1986) have now demonstrated the importance of biological sulphur conversion in forest soils for immobilizing deposited sulphur, although questions pertaining to the rate and period of storage still remain. For example, in grasslands, this form of sulphur storage can be very temporary, since organic sulphur mineralization in surface soils represents the dominant source of sulphur for plant growth (Maynard et al. 1984).

All four western provinces contain soils that are considered to be sulphur-deficient (Terman 1978), and ammonium sulphate is often used as a fertilizer. Ammonium sulphate has also been recorded in areas where liquid manure is sprayed on land surfaces, mainly in winter and early spring (van Breeman et al. 1982). This compound is known to have an acidifying effect on soils. For example, moderately sensitive Grey Luvisol soils in Alberta (typical of the boreal plains ecozone - see Figure 4.3) treated with low doses of ammonium sulphate ($0.6 \text{ keq of S} \cdot \text{ha}^{-1}$) for nearly forty years, showed an increase in soil acidity from pH 6.0 to 5.4 (McCoy and Webster 1977). Although in absolute terms this is a relatively small increase in acidity, from an agronomic standpoint this increase is significant, since at the lower pH value sufficient aluminum is mobilized to be toxic to common crops (Hoyt et al. 1974). Thus, soils in parts of Alberta and northeastern British Columbia having a pH in the range 5.5 to 6.5 are especially sensitive to acidification (Douglas Penney, Alberta Agriculture, pers. comm.) regardless of whether the source of acidity is from the long-range transport of air pollutants, or from fertilizer inputs.

Similar acid sensitivity research has yet to be carried out on prairie soils (e.g., Chernozems), but these are generally considered to be of low sensitivity to acidification (see Figure 4.2). Nevertheless, work done on Solonchak soils (also considered to have low sensitivity to acidification) has shown that even insensitive soils can acidify when ammonium sulphate dosages are high enough. Perl et al. (1982) demonstrated that ammonium sulphate delivered annually over a ten year period, at a rate of $8 \text{ keq S} \cdot \text{ha}^{-1}$, resulted in pH dropping from 5.8 to 4.6. This demonstrates

some of the potential problems associated with trying to extrapolate results from one soil type to another.

A number of important studies carried out in Alberta on the effects of elemental sulphur dust on lodgepole pine forest soils have revealed chemical changes in the surface litter horizons; with the exception of increased sulphur concentrations, though, few changes have been noted in the mineral soil horizons (Maynard 1984). One problem with these studies is that even though the process by which elemental sulphur is oxidized to sulphate in soils is reasonably well known (Burns 1967), we are currently unable to accurately calculate the rate at which oxidation occurs. Consequently, it is extremely difficult to quantify the acid sulphate dose received by systems where the sulphur is initially delivered in elemental form.

In summary, agricultural soils in western Canada are generally less susceptible to atmospheric acidic deposition than are forest soils. In fact, moderate levels of sulphur deposition may be beneficial to many agricultural soils. Sulphate deposition onto forest soils may result in increased leaching of basic cations; at higher elevations, such input may result in aluminum mobilization (due to different soil conditions, base saturation, pH, and other factors). These adverse effects may initially be offset by a fertilization effect. For example, even modest sulphur inputs to a Douglas fir ecosystem in Washington resulted in large excesses in foliar sulphate (Turner et al. 1977). As well, if nitrogen inputs to these systems increase (either through fertilization or atmospheric nitrate deposition), sulphur deficiencies may develop. From studies of forest sulphur requirements for

proper growth (e.g., Evans et al. 1981; MOI 1983), it appears that current deposition levels ($6-10 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$) in western Canada are adequate for forest nutrition. These levels, however, are likely to be inadequate for most crop plants which may require up to $18 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ (Terman 1978).

5.3 Agriculture

The following discussion on the potential effects of acidic deposition on agriculture in western Canada focuses on cultivated crops and does not deal with effects on either grassland plants or associated grazing livestock. The reader should be aware, however, that sulphur exposure can not only directly affect the health of grazing animals (Lillie 1981), but deposited sulphur can also influence the elemental composition of forage plants in ways that may be indirectly deleterious to the animals' health. For example, livestock managers are particularly concerned with sulphur interactions with selenium, copper, and molybdenum and their differential uptake by plants because under conditions of elevated soil sulphur concentrations, deficiency diseases may develop in cattle (Prior et al. 1982a; Prior et al. 1982b). However, deficiency diseases caused by dietary sulphur do not always occur (Kahn et al. 1987).

The remainder of this section includes a summary of the major crops in western Canada and their geographic distribution, as well as a brief review of available dose-response data. To highlight areas of concern, this information is compared to the target loading data.

5.3.1 Agricultural Production in Western Canada

The direct sensitivity of crops to acid-forming pollutants is of primary importance, but the sensitivity of the soils on which the crops are grown is also important. When the map of soil sensitivity (Figure 4.2) is compared to maps of agricultural production (National Atlas of Canada 1974:137-156) it becomes clear that agricultural production is mostly restricted to areas where soils show low sensitivity to acid deposition. There are, however, five areas where soils of moderate to high sensitivity overlap major farming regions:

- 1) the area of Alberta west of Lesser Slave Lake, which is used for growing barley, wheat, and rapeseed (canola), as well as mixed livestock production;
- 2) a region of southwest-southcentral Alberta, used primarily for mixed livestock production, barley, and dairy farming;
- 3) relatively small pockets of highly sensitive soils in central Saskatchewan, where livestock production and wheat are dominant;
- 4) the lower Fraser River valley in southwestern British Columbia; and
- 5) livestock production areas in southcentral British Columbia.

Though liming of agricultural soils is commonly practiced to reduce acidity, this is only practical for intensively farmed areas, and not for rangelands.

5.3.2 Dose-Response Data for Agricultural Plants

The difficulties involved with the interpretation of dose-response data have been critically reviewed (EPA 1984). It is beyond the scope of this section to discuss the experimental design and methods used to determine vegetative responses to acidic precipitation. It should be stressed, however, that data provided by various studies have often been based on different methods making intercomparisons difficult and results appear to be inconsistent. Furthermore, although sulphur dioxide may be the predominant pollutant in a given situation, it is important to recognize that combinations of pollutants may have different effects. Unfortunately, the available data concerning effects of mixtures or sequences of air pollutants on plants are fragmented and there are major gaps in our present knowledge of pollutant interactions (Runeckles 1984). For the following discussion to be relevant and useful, data from various studies were summarized and effects are discussed in relation to a broad range of exposure to oxides of sulphur and nitrogen.

The crops of concern east of the Rocky Mountains are wheat, barley and other cereals, and grasses used for animal feed. While cereals are also the major crops in British Columbia, additional attention should be given in this province to apples, other fruit trees, and vegetables, particularly in the Okanagan and Fraser Valleys.

Three tables in Appendix III, based on several comprehensive literature reviews (EPA 1982a; EPA 1984), provide dose-response data for various field and

horticultural crops. Data from both field studies and investigations under controlled environments are listed together but attention should be given to the advantages and shortcomings of the methods used and their effects on the results (EPA 1984).

The following discussion focuses on the major crops in western Canada.

Cereals

Field studies on wheat have shown that sulphur dioxide concentrations up to $1,180 \mu\text{g}\cdot\text{m}^{-3}$ had no effect on yield when exposure was for 3 hr and up to eight times during the growing season. Shorter exposure to higher concentrations (up to $31,400 \mu\text{g}\cdot\text{m}^{-3}$) also had no effect. Lower yield (by 42%), however, was found with Durum wheat when exposed to $390 \mu\text{g}\cdot\text{m}^{-3}$ for 72 hr per week for the growing season. Spring wheat was not affected by the latter conditions (EPA 1982a).

Injurious effects of nitrogen dioxide on wheat have been reported (EPA 1982b) with concentrations of 28.5 and $38.0 \text{ mg}\cdot\text{m}^{-3}$ for 1 and 0.5 hours, respectively. No effects were seen at lower concentrations (18.8 or $10 \text{ mg}\cdot\text{m}^{-3}$) even under extended exposure (up to 7 hours). Yield studies with $2 \text{ mg}\cdot\text{m}^{-3}$ showed no effect on grain yield but the straw yield was reduced by 12% (EPA 1982b). Oats were injured by nitrogen dioxide when exposed to $18.8 \text{ mg}\cdot\text{m}^{-3}$ or higher concentrations for several hours but showed high cultivar variability. None of the corn cultivars tested were affected at these concentrations (EPA 1982b).

Controlled environment studies with sulphate and nitrate at simulant concentrations up to $98 \times 10^6 \text{ g}\cdot\text{m}^{-1}$ and $74 \times 10^4 \text{ }\mu\text{g}\cdot\text{m}^{-3}$, respectively, showed no effect on market yield of "Fieldwin" wheat (EPA 1984a). The total deposition under the exposure conditions was up to 450 and $3.4 \text{ kg}\cdot\text{ha}^{-1}$ for the oxides of sulphur and nitrogen, respectively. The pH range during the study was 5.6 to 3.0.

Data from field research are also available for corn. A 9% decrease in yield resulted from treatments with 42 times the ambient hydrogen ions deposition (pH 4.0), but no effect occurred at 132 and 187 times (pH 3.5 and 3.0, respectively) (EPA 1984a). The respective loadings of sulphate and nitrate at pH 3.0 were 198 and $135 \text{ kg}\cdot\text{ha}^{-1}$, and at pH 4.0, 19 and $17 \text{ kg}\cdot\text{ha}^{-1}$ respectively. These non-linear dose-response relationships were common among crops that showed susceptibility to only one concentration (EPA 1984a). The reasons for these results are not clear and indicate that involvement of other parameters, such as the ion ratio of sulphate to nitrate or effect on seed germination and seedlings establishment, should not be ruled out. Experiments with grasses (Fescue) have shown up to 24% yield increase when the ratio of sulphate to nitrate was 1:6.4 (pH 4.0), but no effect has been reported at the same pH where the sulphate:nitrate ratio was 1:1. Studies with several grasses (EPA 1982b) have shown that growth inhibition effects of nitrogen dioxide and sulphur dioxide are synergistic.

Controlled environment studies on barley and oats have found that neither crop was affected by simultaneous deposition of sulphate and nitrate of up to 450 and $3.3 \text{ kg}\cdot\text{ha}^{-1}$, respectively (pH range of 5.6-3.0). In another

study (EPA 1980), grasses and corn showed slight foliar injury when treated with sulphuric rain (pH 3.0), but no effect was observed with other cereals including wheat and barley.

Apples and Other Horticultural Crops

Exposure for 6 hr to increasing concentrations of sulphur dioxide (from 1,050 to 6,550 $\text{mg}\cdot\text{m}^{-3}$) has been associated with up to 62% increase in leaf abscission. No effect on yield has been recorded (EPA 1982b). Information for other fruit trees is not available.

Vegetables and Field Crops

Cultivar and methodology differences do not allow for any conclusive discussion. Yet, some crops can be classified based on trends in sensitivity observed in field research. Negative yield effect at all acidity levels (pH 3.0, 3.5, and 4.0) was shown in garden beet (EPA 1984). Mustard and several soybean varieties were adversely affected by at least one of the above acidity levels, and radish, alfalfa, potatoes, and other cultivars of soybean had higher yields when treated with acid deposition (EPA 1984). The majority of many other crops including cultivars of spinach, various beans, soybean, radish, alfalfa, and mustard green were not affected by these concentrations of sulphur and nitrogen oxides. Sunflower's yield increased when plants were exposed to 160 - 1,390 $\text{mg}\cdot\text{m}^{-3}$ of sulphur dioxide for 24 hours, but yield decreased (up to 41%) and leaf injury (up to 16%) was evident at exposure conditions of 790 $\text{mg}\cdot\text{m}^{-3}$ of sulphur dioxide for 5 hours per day over 12 weeks. The yield of pea

was reduced (5%) following exposure to $260 \text{ mg} \cdot \text{m}^{-3}$ of sulphur dioxide for 18 days, and to a larger extent (26% reduction) at $660 \text{ mg} \cdot \text{m}^{-3}$ of sulphur dioxide. Dose-response data for other crops in the region including sugar beet, flaxseed, and rapeseed (canola) are not available.

5.4 Forests

5.4.1 Introduction

The effects of acidic deposition on forests have been the subject of numerous reviews (e.g., Smith 1981; Fernandez 1983; David et al. 1984; Bennett et al. 1985) and conference proceedings (e.g., Northeastern Forest Experiment Station 1976; Hutchinson and Havas 1980; Drablos and Tollan 1980; Miller 1980; Acid Rain Foundation 1985; FRNRF 1985). This section:

- 1) examines the distribution of major tree species in western Canada (the resource potentially at risk);
- 2) provides a detailed discussion of the existing evidence for potential impacts from acidic deposition on these particular species;
- 3) places acidic deposition in context with other forms of pollution harmful to forests; and
- 4) contains a discussion of the adequacy of existing information for the purposes of establishing target loadings in western Canada.

There are at least seven genera of coniferous trees and at least three genera of deciduous trees that are considered to be major important species in western Canada. Some, like both the white and black spruces, tamarack, white birch, trembling aspen, and willows (not truly a tree), are found in all six provinces while others, such as the western red cedar and western hemlock, are found only in single provinces (Table 5.2a). The trees of western Canada have been divided into six major assemblages (Hosie 1969). While individual species of spruce, fir, and pine are generally associated with only one biome, Douglas fir, cedar, hemlock, and nearly all deciduous species are included in two or more biomes (Table 5.2b). In addition to the major tree species listed in Table 5.2, each biome also contains many other plant and animal species that together constitute a functional ecosystem.

If target loadings were prescribed for a specific biome, then presumably the objective would be to protect the structural and functional integrity of that entire ecosystem. To achieve this, the target loading must permit maintenance of not only the forest's biological integrity, but also the chemical and physical characteristics as well. This raises the difficult question of what are the most definitive indices of ecosystem response to pollution (see Odum 1985). Notwithstanding these complexities (and those associated with pollutant interactions), the decision was made to evaluate dose-response data for individual western Canadian tree species since relatively little is known of the intrinsic responses of ecosystems to atmospheric pollutants in this part of the country.

Table 5.2
Major forest tree species in western Canada by
province a) and biome b)

a)

SPECIES	PROVINCE (1)					
	BC	A	S	M	Y	NWT
<u>CONIFEROUS</u>						
White Spruce	x	x	x	x	x	x
Black Spruce	x	x	x	x	x	x
Engleman Spruce	x	x				
Sitka Spruce	x					
Balsam Fir		x	x	x		
Alpine Fir	x	x			x	
Jack Pine		x	x	x		x
Lodgepole Pine	x	x			x	x
Ponderosa Pine	x					
Douglas-Fir	x	x				
W.* Red Cedar	x					
W. Hemlock	x					
Tamarack	x	x	x	x	x	x
<u>DECIDUOUS</u>						
White Birch	x	x	x	x	x	x
Trembling Aspen	x	x	x	x	x	x
Willow	x	x	x	x	x	x

(1) BC = British Columbia; A = Alberta; S = Saskatchewan; M = Manitoba; Y = Yukon; NWT = Northwest Territories; W = western

b)

SPECIES	BIOME (2)					
	B	SA	M	CB	CT	G
<u>CONIFEROUS</u>						
White Spruce	x					
Black Spruce		x				
Engleman Spruce						
Sitka Spruce					x	
Balsam Fir	x					
Alpine Fir		x				
Jack Pine	x					
Lodgepole Pine		x	x			
Ponderosa Pine			x			
Douglas-Fir			x	x	x	
W.* Red Cedar				x	x	
W. Hemlock				x	x	
Tamarack	x					
<u>DECIDUOUS</u>						
White Birch	x					
Trembling Aspen	x		x			x
Willow	x					x

(2) B = Boreal; SA = Subalpine; M = Montana;
CB = Columbia; CT = Coastal; G = Grassland;
W = western

Source: Adapted from Hosie 1969.

5.4.2 Atmospheric Pollutant Dosage

Despite the availability of numerous studies on responses of tree species to air pollutants, few of these studies provide data that are directly relevant to the target loading issue. This is largely because few of these studies have been specifically designed to carefully document the dose of atmospheric pollutants. Furthermore, while some of the studies cited below were carried out along a deposition gradient, covariation of gaseous pollutants, sulphate, acid, and metals makes it difficult to extrapolate results to other locations. As well, failure to clearly identify either the causative agent or effective dosage makes it difficult to apply these data to policy formulation. In addition, pollutant episodes can complicate matters, especially when describing dosages. Consequently, when dosages are being discussed in the next section, the reader must bear these limitations in mind.

5.4.3 Vegetative Responses to Atmospheric Pollutants

There are several sites in western Canada where terrestrial ecosystem responses to atmospheric pollutants have been investigated. These include studies around the smelters in Thompson (Wotton et al. 1985) and Flin Flon, Manitoba (Hogan and Wotton 1984), the oil sands plant in northern Alberta (Smith 1981; Addison 1982), natural gas processing sites in Alberta (Sandhu et al. 1980; Legge 1980; Allen et al. 1982; Legge and Bogner 1982; Addison et al. 1984a) and in British Columbia (Kotturi et al. 1985), and the historic work around the smelter in Trail, British Columbia (National Research Council of Canada 1939; Scheffer and Hedgcock 1955). Taken together, these studies reveal:

- 1) certain native, non-vascular plants such as mosses (Kennedy et al. 1985) and lichens (Addison 1984c) are sensitive to increased exposure to sulphur in either elemental (particulate) or oxidized form (gas or liquid, or both), and can be useful as bioindicators (Case, circa 1981);
- 2) most forest understory vegetation appears to be relatively unaffected by increased sulphur deposition, except under extreme loadings (Addison et al. 1984a; Wotton et al. 1985); and
- 3) in some cases (using certain indicators), deciduous woody species in western Canada appear to be more sensitive to air pollutants than coniferous species (e.g., Addison et al. 1984b), while other studies have shown that evergreen foliage in boreal forest foliage is less well buffered against acidic deposition than that of broad-leaved species (Pylypec and Redmann 1984).

The geographical location of emissions sources relative to the biogeographic distribution of forest species has resulted in efforts being concentrated on a relatively small number of species including jack pine (Pinus banksiana), lodgepole pine (Pinus contorta), white spruce (Picea glauca), black spruce (P. mariana), aspen (Populus tremuloides), willow (Salix spp.), alder (Alnus spp.), and paper birch (Betula papyrifera). As well, data pertaining to the sensitivity of some western Canadian forest species are available in the scientific literature based on studies outside this region. For example, responses of ponderosa

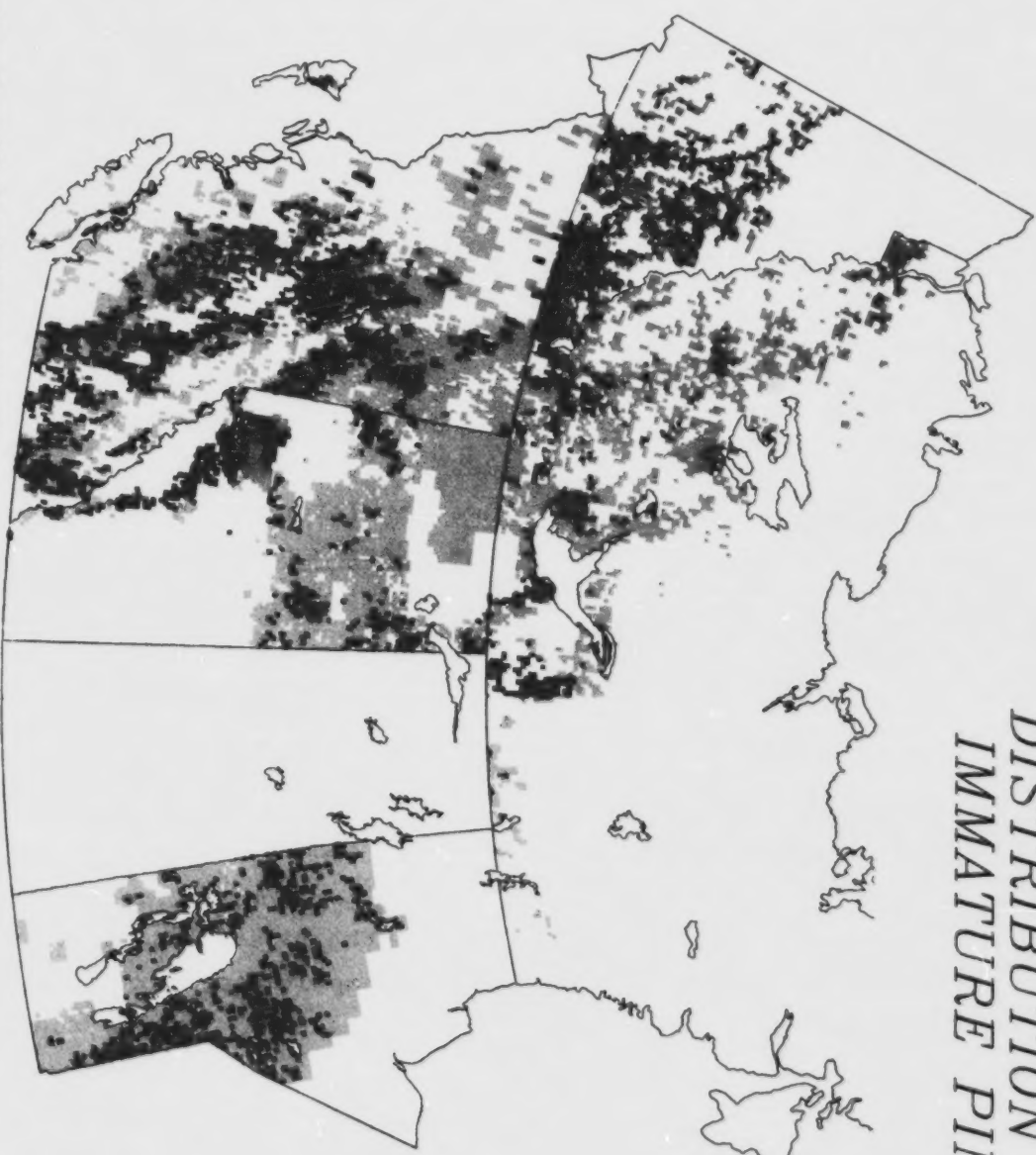
pine (Pinus ponderosa) to elevated ozone have been studied in California (e.g., Williams et al. 1977); similar data (San Bernadino Valley) have been used to quantify economic damage associated with air pollutant damage (Westman 1977).

Limited laboratory experimentation has been carried out on germination and seedling growth of tree species found in western Canada when exposed to various air pollutants. Again, it appears that deciduous species such as B. papyrifera and P. tremuloides are more sensitive than conifers like P. banksiana (Cox 1982). Furthermore, studies have shown that early growth in some coniferous species, for example Douglas fir (Pseudotsuga menziesii), actually improves with mild acid treatments down to pH 3.0 (Lee and Weber 1979). Unfortunately, the significance of such laboratory results for survival and early growth under field conditions has yet to be determined.

Assuming that immature members of a species' population have equal or greater susceptibility to damage from atmospheric pollutants, it is reasonable to examine the spatial overlap between pollutant sources and the distribution of "sensitive" plants. Data on the geographic location and density of different forest species are available through the FORSTATS program of the Canadian Forestry Service (CFS). For example, distribution and density of immature pine in western Canada are illustrated in Figure 5.2. Comparing this with Figure 3.2, it can be seen that immature pine forests in the Alberta foothills coincide with a region of major sulphur oxide emissions from natural gas processing. As well, this species appears to be located mainly in areas having moderate to low soil sensitivity (Figure 4.2) and moderate to high

CANADA

DISTRIBUTION OF IMMATURE PINE



% volume by cell

0 or Undetermined
>0 - 5
>5 - 10
>10 - 20
>20

0 150 300 km
1:15 000 000

Reference: Bonner, G.M., Canada's Forest Inventory 1981.
Footnote: For Saskatchewan age class is undetermined.
No species differentiation of hardwood and softwood volumes reported for Yukon and Northwest Territories.
PRODUCED BY THE PETAMAWA NATIONAL FORESTRY INSTITUTE, CANADIAN FORESTRY SERVICE, 1986.

Figure 5.2: Distribution and density of immature pine.

ability to neutralize incoming acidity (Figure 4.1). This suggests that atmospheric pollutants are most likely to affect immature pine trees through direct vegetative impacts, rather than through changes in the chemistry and nutrient status of underlying soils. This simple example illustrates how such analyses can be performed using species-specific data on distribution and density; the reader is, however, cautioned that individuals classified as immature by the Canadian Forest Service may not be the same as the seedling stage used by many researchers in experimental laboratory studies.

Overall, it appears that current levels of atmospheric sulphur deposition in western Canada are mainly responsible for localized effects, at least over the time periods for which most studies are available (<15 years). Relative insensitivity of forest species to low levels of sulphur compounds and acidity is consistent with current thinking by Cowling (1984) and Aber (1985), both of whom rank ozone and nitrogen compounds as greater threats to forest ecosystems. Cowling's ranking, in order of decreasing importance, is as follows:

- 1) ozone;
- 2) total biologically available nitrogen compounds, including all forms of nitrates, ammonium, and ammonia;
- 3) phytotoxic gases (e.g., nitrogen oxides, sulphur dioxide);
- 4) toxic metals;

- 5) nutrient-and acidity-determining ions (e.g., acid precipitation); and
- 6) growth altering organic chemicals such as ethylene and aniline (aminobenzine).

It is also possible that acids deposited through fog, light drizzle, or evaporation events may rank higher on the scale than acids deposited directly from precipitation.

In a recent article, McLaughlin (1985) ranked the sensitivity of several tree species found in western Canada to various air pollutants; his rankings are presented in Table 5.3. From examining the information in this table, and comparing it with Table 5.2, the reader can see that:

- 1) relatively little is known about the sensitivity of most trees in western Canada to most of the major atmospheric pollutants;
- 2) considerably more is known about the sensitivity of boreal forest trees than is known about species found in other western forest biomes;
- 3) coniferous species appear to be somewhat more tolerant to atmospheric pollutants than deciduous species; and
- 4) proportionally less is known about the sensitivity of major tree species in British Columbia than in other western provinces.

Table 5.3

Susceptibility of tree species to selected air pollutants

Forest Tree Species	POLLUTANTS OF CONCERN				
	Sulphur Oxides	Nitrogen Oxides	Photochemical Oxidants	Particulate Matter	Organics
<u>CONIFEROUS</u>					
White Spruce	5,6	0	6,7	0	0
Black Spruce	5,6	0	0	0	0
Engleman Spruce	0	0	0	0	0
Sitka Spruce	0	0	0	0	0
Balsam Fir	4	0	6	0	0
Alpine Fir	0	0	0	0	0
Jack Pine	2,5	0	2	0	0
Lodgepole Pine	0	0	0	0	0
Ponderosa Pine	0	0	3,4	0	0
Douglas-Fir	3,4	0	5,6	0	0
W. Red Cedar	0	0	0	0	0
W. Hemlock	0	0	0	0	0
Tamarack	0	0	0	0	0
<u>DECIDUOUS</u>					
White Birch	2	0	0	0	0
Trembling Aspen	2	0	2,3	0	0
Willow	2	0	0	0	0

Source: Adapted from: Addison et al. 1984b; McLaughlin 1985.

0 = not rated
1 = most sensitive
7 = most tolerant

5.4.4 Target Loading Considerations

From the foregoing we can conclude that despite the large and rapidly growing body of literature pertaining to vegetative impacts resulting from atmospheric pollutants, there are still inadequate data to support development of scientifically defensible target loadings in western Canada. This is not surprising; David et al. (1984) were unable to identify a "threshold" concentration of acidic deposition below which forest damage would be prevented even in the more thoroughly studied forests of the northeastern United States. As Bormann (1985: p.439) has pointed out "even though terrestrial ecosystems in large regions may be, and probably are, responding to air pollution stress, we cannot yet measure these responses adequately."

In preparing to establish target loadings for acidic deposition in Minnesota, a comprehensive study was undertaken in which the authors concluded, based on the available evidence, that target loadings capable of protecting sensitive aquatic ecosystems would probably be adequate to protect the long-term integrity of forested ecosystems (Minnesota Pollution Control Agency 1985). Considering these findings, the complementary conclusions of Evans et al. (1981), and recognizing that the paucity of data regarding sensitivity of major western Canadian tree species to atmospherically-deposited sulphur makes it virtually impossible to develop defensible target loadings for this ecosystem at the present time, we suggest that target loadings designed to protect sensitive aquatic ecosystems in western Canada be used for the time being to also provide some protection to forest ecosystems in the same regions. To move beyond these

interim target loadings to defensible criteria for the protection of forests, further research and data analysis is required. The recommended research activities are described in Section 7.

In areas considered to have highly sensitive soils, but not having surface waters of high sensitivity, the establishment of scientifically defensible target loadings is more difficult. This issue is discussed further in Section 6.

5.5 Aquatic Systems

5.5.1 Introduction

Over the last fifteen years, a tremendous amount of research has been conducted on the effects of acidic precipitation on aquatic systems. It is our intention here to focus on only a thin slice of this research, namely the syntheses and analyses which provide direction to establishing dose-response relationships and target loadings. Following this introduction, which sets up a conceptual framework, this section reviews the approaches available to defining dose-response relationships for aquatic systems. There are several detailed reviews of aquatic system acidification research, covering Norwegian acidification studies (Overrein et al. 1980), the effects of acidification on aquatic systems (Harvey et al. 1981; Haines 1981; Environmental Protection Agency 1984), the phenomenon of acidic episodes (Marmorek et al. 1984, 1986), the role of organic acids (Jones et al. 1986), and the factors affecting the rate of acidification of surface waters (Church and

Turner 1986). These references should be consulted by those readers requiring more detail.

Though an enormous amount of information is available, the definition of dose-response relationships for aquatic systems nevertheless raises some fundamental problems:

- 1) How should "aquatic systems" be defined?
- 2) How should the response of aquatic systems be represented?
- 3) How should the dose to aquatic systems be represented?

For the purposes of this document, aquatic systems are defined as streams or lakes, the biota contained therein, and their contributing catchments. This definition works well for all surface waters except seepage lakes, which have ill-defined catchments.

How should the response of aquatic systems be represented? Evidence exists showing responses to acidic deposition in soil water chemistry, surface water chemistry, and many different forms of aquatic biota. Measuring the responses of soils and soil water is difficult, primarily because of the tremendous spatial variability of soils within each catchment. Average lake alkalinity or pH is the most easily measured, and therefore the most commonly used measure of surface water chemical response. There are, however, two very strong reasons for estimating response in terms of changes in aquatic biota, rather than surface water

chemistry. First, the responses of biota ultimately integrate, over both space and time, the many complex processes occurring in surface waters. Second, in protecting aquatic ecosystems one is ultimately most concerned about the health of aquatic biota (particularly the most economically important organisms such as fish) with a secondary interest in the quality of the habitat supporting these biota. Therefore, the response of aquatic systems should ideally be measured as changes in the biotic variables considered to best represent the state of health of aquatic biota, including such population indicators as species presence (or absence), yield, or recruitment, and community measures such as species diversity, biomass, and production.

While recognizing the primacy of biological responses, researchers attempting to develop dose-response relationships have chosen a variety of approaches:

- 1) compare the state of health of aquatic biota in regions with different deposition levels;
- 2) compare biologically relevant surface water chemistry conditions (pH and alkalinity, aluminum concentrations) in regions with different deposition levels; and
- 3) disaggregate the system into two components (deposition ---> surface water quality response, and surface water quality ---> biological response), and develop functional relationships (or models) for each component.

The first two approaches are purely empirical. That is, they do not consider watershed processes or the sensitivity of aquatic ecosystems. These approaches are described together in Section 5.5.2, "Empirical Comparisons." The third approach involves some consideration of watershed processes and aquatic system sensitivity, though the level of detail with which these factors are represented varies with different models. Since this approach involves disaggregating the system into two parts, the discussion is presented in two sections: Section 5.5.3, "Water Quality Thresholds for Aquatic Biota"; and Section 5.5.4, "Linking Deposition Levels with Water Quality Thresholds."

How should the dose to aquatic systems be represented? This question is discussed in more detail in the following sections with respect to particular models. In general, most discussions of dose-response or target loadings consider the dose in terms of the annual wet deposition of sulphate or the annual mean pH of precipitation, primarily because these are easily measured quantities. Though practical, these definitions of dose ignore three important factors: dry deposition, alkaline particulates, and seasonal pulses in loadings. A better definition of "total effective acid deposition" (suggested by Scheider and Dillon 1982) recognizes the acidifying effect of dry deposition of gaseous sulphur, and wet deposition of ammonium, as well as the acid neutralizing effect of nitrate deposition:

$$\text{Acidic Deposition} = (D)H^+ + (D)SO_2 + (D)NH_4^+ - k * (D)NO_3^-$$

where: $k = 1$ for aquatic systems;

$k < 1$ for terrestrial systems; and

(D) = deposition of the acid-forming substances
specified in each part of the equation.

Though the approach of Scheider and Dillon is worth pursuing in the long term, in western Canada estimates of dry (SO_2) deposition are scarce further from specific point sources. Also, the above definition does not take into account the additional correction necessary within prairie regions to account for alkaline particulates. Section 2 of this report showed that wet sulphate concentrations are positively correlated with pH for many CANSAP stations in western Canada (see Figure 2.3), indicating that for these stations wet sulphate deposition is a poor indicator of acid loading. It is critically important to determine which CANSAP stations in western Canada show these effects of alkaline particulates, and which behave like stations in eastern Canada.

Since the areas of greatest aquatic sensitivity are generally located away from prairie regions (presumably the largest dust sources), developing a correction for alkaline particulates may not be a practical concern. For example, the Vancouver CANSAP stations do not show the effects of alkaline particulates. It is important, however, to examine to what extent stations in other regions of sensitive surface waters (e.g., northern Saskatchewan, parts of northern Alberta) are influenced by this phenomenon. If a correction for alkaline particulates is required, it promises to be more difficult to derive than seasalt corrections. Seasalt corrections rely on the general constancy of seawater composition; soil composition is considerably more variable.

Episodic declines in surface water pH during snowmelt and storm events can significantly impair the habitat of aquatic biota (reviewed by Marmorek et al. 1984, 1986). A measure of the dose received by aquatic systems during such

periods (e.g., mean annual maximum monthly load of acids in deposition plus meltwater) is much more difficult to define, but would be a useful supplementary measure of dose.

5.5.2 Empirical Comparisons

As mentioned in the above introduction, empirical comparisons include two approaches: comparisons of deposition with biological responses, and comparisons of deposition with surface water chemistry. An example of each is included here.

Newcombe (1985) synthesized the results of various intensive studies into a hierarchy of harmful biological effects at different levels of sulphate deposition (Figure 5.3). This figure provides a useful overview of the range of observed responses to sulphate deposition within relatively sensitive aquatic systems. Newcombe proposed a limit of between 10 and 20 kg of sulphate $\cdot$ ha $^{-1}\cdot$ yr $^{-1}$ for the protection of aquatic systems.

The State of Minnesota Pollution Control Agency has recently prepared a comprehensive review of information relevant to the establishment of target loadings (Minnesota Pollution Control Agency 1985). Included in their review was a summary of empirical observations adapted from Brydges and Neary (1984), which has been reproduced in Table 5.4. This table also indicates an increase in damage to aquatic habitat or fisheries above 20 kg of sulphate $\cdot$ ha $^{-1}\cdot$ yr $^{-1}$. The regions shown in Table 5.4 vary considerably in the level of annual precipitation received, which affects the estimated target loading, as discussed in Section 6.

Table 5.4

Summary of empirical observations of aquatic regime response to sulphate deposition in specific study areas. Adapted from Brydges and Neary (1984), cited in Minnesota Pollution Control Agency (1985)

Location	SUMMARY	
	Deposition kg sulphate·ha ⁻¹ ·yr ⁻¹	Effects
Northern Saskatchewan	5 wet (1980)	No chemical effects.
Kenora, Ontario	9-11 bulk (1972)	No chemical effects.
Minnesota	10-15 wet (1980)	No chemical effects.
Northern Wisconsin	16-17 wet (1981)	Some acidification (Nichols and Verry 1985).
Algoma, Ontario	24.7 wet (1981)	pH depression of 2.1 units; Elevated excess sulphate relative to region not receiving acidic deposition; More lakes of low pH than expected.
Nova Scotia	22 wet (1977-81)	Loss of Atlantic salmon species; Historic record of decreased pH in rivers.
Laurentide Park, Quebec	22-40 wet (1977-80)	Indication of decreased pH in some lakes; indication of decline in angling success in lower pH lakes; lower pH in lakes in spring than in summer.
Adirondacks, NY	32-40 wet (1978) 29 wet (1980)	Evidence of pH declines and loss of fish populations over time.
Maine	17-28 wet	Evidence of slight pH decrease in lakes (historic records); no effects on Atlantic salmon; no evidence of effects on fish in inland lakes.
Hubbard Brook, NH	36 wet (1981) 22 wet (1980)	Spring pH depressions; no long-term change in stream or lake pH.
Muskoka-Haliburton, Ontario	23-29 wet (1976-78) 31-42 bulk	pH depressions; fish kill associated with pH depression in one lake; algal composition in lakes related to pH.

Though only preliminary survey information is available, the existing evidence does not indicate significant water quality impacts from acid deposition in western Canada. Intensively studied watersheds in Haney, British Columbia, just east of Vancouver, receive acidic rain of pH 4.7, and a total sulphate loading of about 18 kg of sulphate per hectare per year (Feller and Kimmins 1979, 1984). During the period from 1972 to 1978, these relatively low alkalinity streams ($170 \mu\text{eq}\cdot\text{L}^{-1}$) showed no significant trends in alkalinity (Feller and Kimmins 1984). This level of loading is also not associated with acidification in northern Saskatchewan (Table 5.4, Liaw and Atton 1981; Shewchuk 1983). To assess whether coastal systems in British Columbia are more tolerant, one would need to monitor systems receiving doses of about 20 kg of sulphate $\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ (either from point sources or experimental manipulations).

Other data also suggest that existing levels of deposition are probably not causing extensive changes in water quality or fish communities in British Columbia. Survey data in this province indicate that only 9 of 700 lakes have pH levels less than 6.0 (Swain 1985), and there is no evidence of fish loss in these lakes. Of these nine lakes, only four had colour data, and only one of these four was highly coloured (L.G. Swain, Ministry of Environment, Victoria, British Columbia, pers. comm.). A synoptic survey of 174 Pacific salmon spawning streams in British Columbia and the Yukon indicated that over half (90) had alkalinities less than $200 \mu\text{eq}\cdot\text{L}^{-1}$ for at least part of the year (Sullivan et al. 1985). Sixteen streams (9% of the population sampled) had pH levels less than 6.0, though five of these sixteen exhibited marked colouration, suggestive of organic rather than mineral acids. More intensive monitoring of the eleven

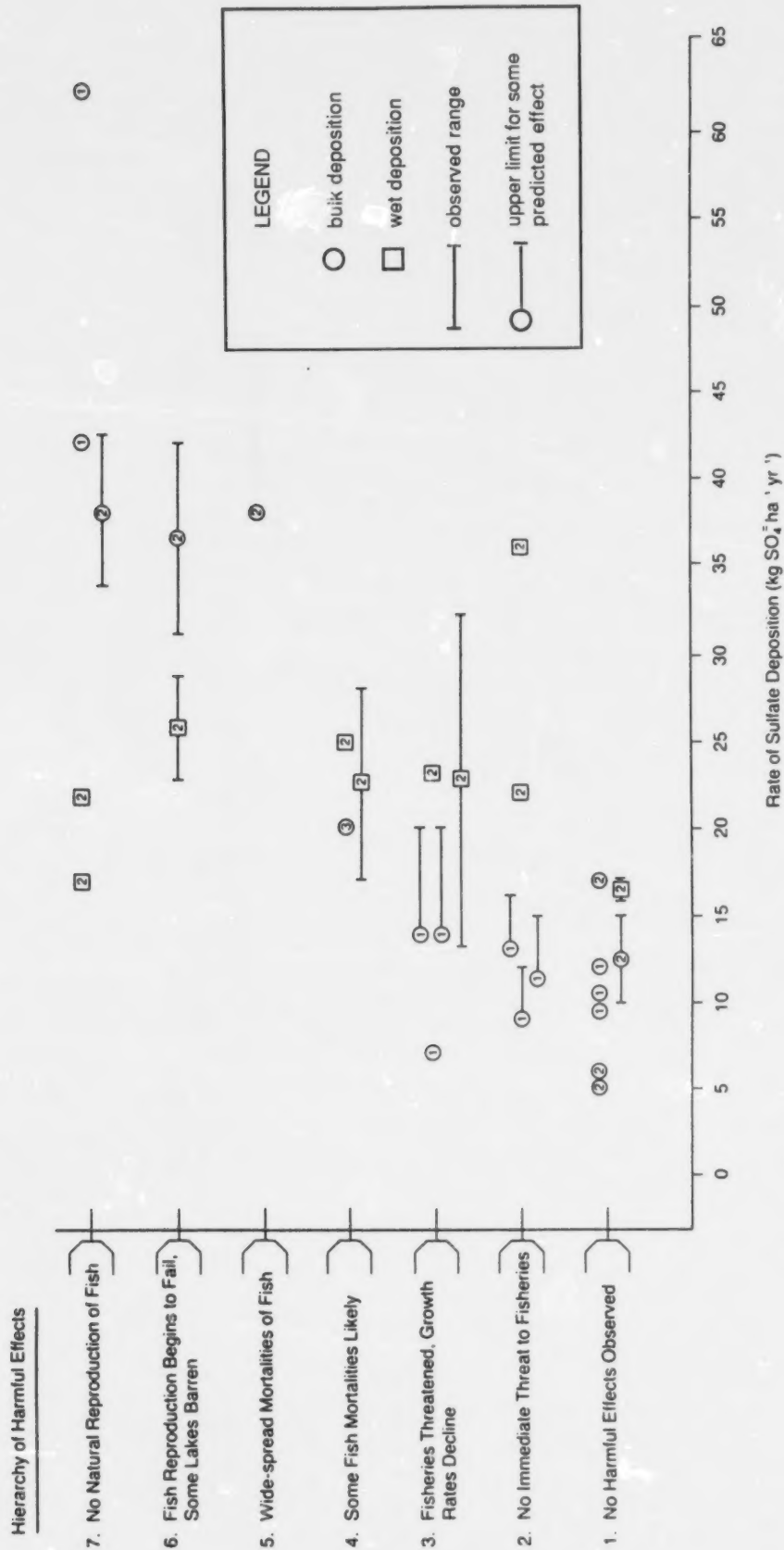


Figure 5.3: A hierarchy of the harmful effects of acid deposition as a function of deposition rate in eastern North America. 1, Glass and Brydges (1982); 2, U.S.-Can. MOE (1983); and 3, Harvey et al. (1981). Source: Newcombe (1985).

non-coloured streams is necessary to ascertain why their pH levels were less than 6.0. It is possible that a greater proportion of streams exhibit acidic episodes, since sampling was not sufficiently intensive to detect these incidents.

There are several difficulties associated with directly comparing deposition levels in different regions with observed biological responses. First, an accurate description of biological response within a region necessitates measurements of response on a large number of systems, so that the regional variability in aquatic systems is captured. Measurements of most biotic response variables (with the exception of presence/absence) require intensive studies, which, due to budget limitations, can only realistically be performed on a few systems. One is therefore confronted with the trade-off between reasonably crude indicators (such as presence/absence) for a large number of systems versus more sensitive indicators of biological response for a few systems that may not be regionally representative. Empirical comparisons of deposition and water chemistry do not share this problem, since it is relatively easy to measure indicators of surface water chemistry for a large number of systems.

The second problem with comparisons of deposition rates and the resulting biological responses is their lack of transferability. Intensive studies of biological changes over time, together with extensive studies of fish distribution (presence/absence), have been performed in the La Cloche Mountain and Sudbury regions of Ontario, the Adirondack region of New York state, Nova Scotia, and southern Norway and Sweden (reviewed in Harvey et al. 1981,

and Baker 1984). Indeed, these areas provide some of the strongest evidence for the potential impacts of acidic deposition. The difficulty lies in transferring these results to other regions, with different deposition levels, deposition composition, climates, geology, soils, and aquatic biota. Simple empirical comparisons of deposition and water chemistry are also difficult to transfer from one region to another. They lack the structure provided by quantitative models which can be developed in one region and formally tested in another. These models are discussed in Section 5.5.4.

5.5.3 Water Quality Thresholds for Aquatic Biota

The decline of fisheries in Scandinavia and North America has focused attention on the water quality conditions associated with changes in the reproduction, growth, and survival of fish. Though fish are of primary economic importance, a large number of other biological changes can occur in aquatic systems prior to the elimination of fish. This is demonstrated by Figure 5.4, which shows the relative number of taxa of major taxonomic groups associated with different pH levels (Eilers and Berg 1982, cited in MOI 1983). This figure, however, only demonstrates a correlation between pH levels and numbers of taxa. A more convincing illustration of the variety of ecosystem changes caused by acidification is shown in Table 5.5, which summarizes the results of the experimental acidification of Lake 223 in the Experimental Lakes Area (ELA) of northwestern Ontario. Both Figure 5.4 and Table 5.5 indicate incipient changes in aquatic systems at pH levels of 5.8 - 6.0.

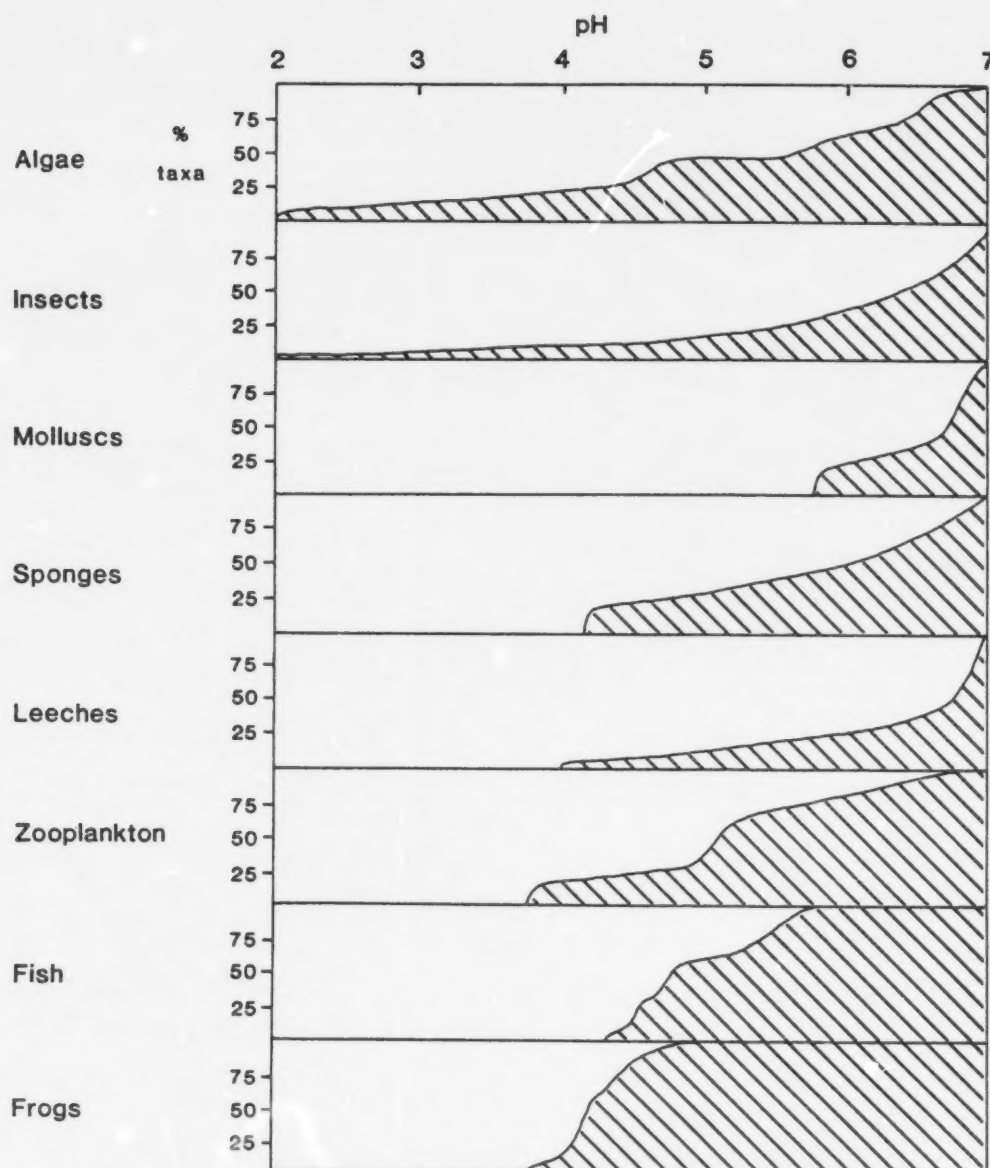


Figure 5.4: Relative number of taxa of the major taxonomic groups as a function of pH (Eilers and Berg 1982), cited in MOI (1983).

The experiments conducted at the ELA have special relevance to the areas of Manitoba and Saskatchewan considered to be sensitive to acidic deposition (see Section 4), since both the terrain and fish communities are similar to those found at the ELA. Few other acidification experiments have been conducted in western Canada, but those which have been, suggest that the aquatic biota are similar in sensitivity to the biota of eastern Canada. Acidification experiments in a coastal mountain stream in British Columbia produced significant escape responses in aquatic insects at pH 5.9 (Bernard 1985). Acidification of enclosed zooplankton communities caused major changes in species composition at pH 5.4 (Marmorek 1984).

Though maintenance of the entire aquatic ecosystem is the primary concern, much research attention has been focused on fisheries, both because of their economic importance and their relative sensitivity to acidification effects. Much of the vast literature concerning acidification effects on fisheries was elegantly synthesized by Baker (1984) in a single figure, which is included here as Figure 5.5. Though data for some species are very limited (e.g., only one reference for some species), declines in reproduction and abundance occur for the most sensitive species in the pH range 6.0 - 5.5, and the lower limit for persistence appears to be 4.5. It should be noted that the pH threshold for any species is dependent not only on pH, but also on the concentrations of both inorganic monomeric aluminum and calcium. Furthermore, most of the data used to construct Figure 5.5 were obtained from laboratory experiments, which cannot be directly extrapolated to the field (Baker 1984). Empirical relationships, based on the presence or absence of

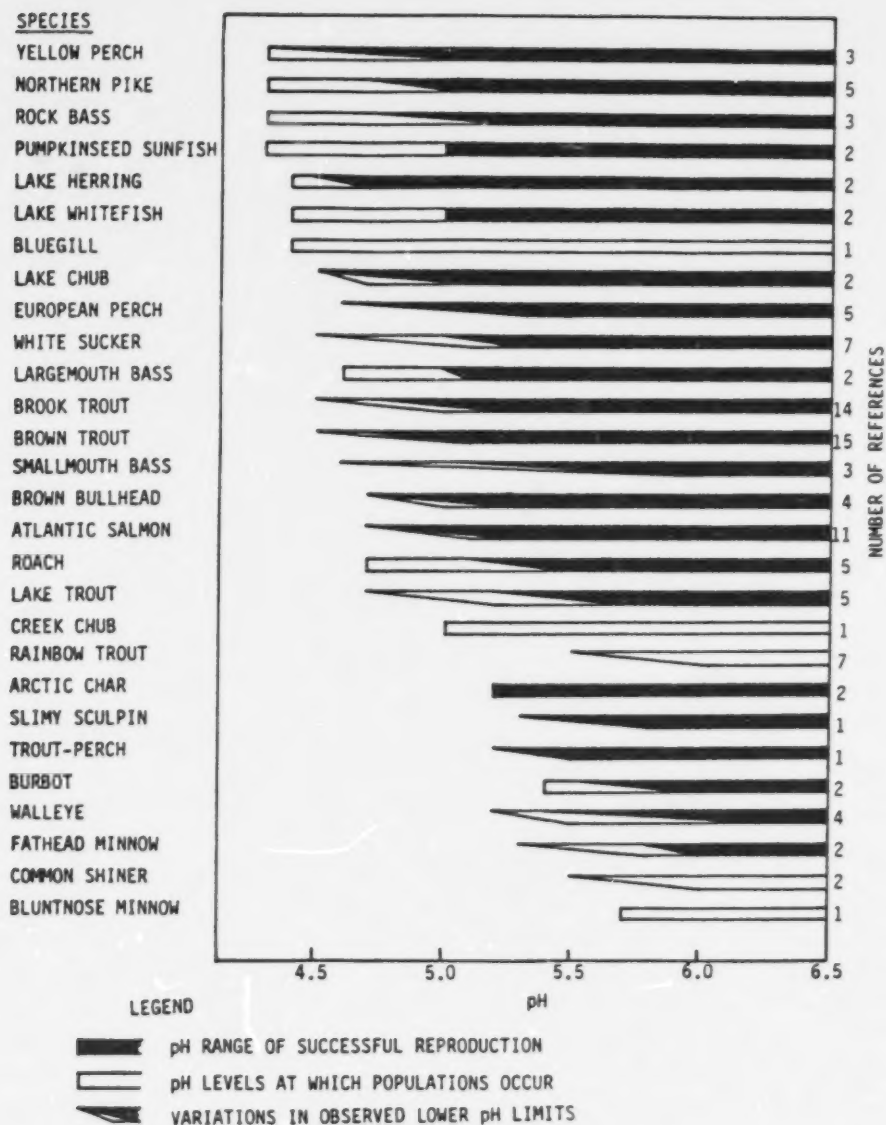


Figure 5.5: Initial estimates of relationship between acidity and fish response. Ranges should be treated cautiously, since some species have only received limited study, or the studies completed did not consider effects on reproduction (indicated by an asterisk). Source: Baker (1984).

Summary of results of experimental acidification of Lake 223 in northwestern Ontario. (Source: Schindler, D.W. et al. 1985)

Acidification Regime 1974 - 1983

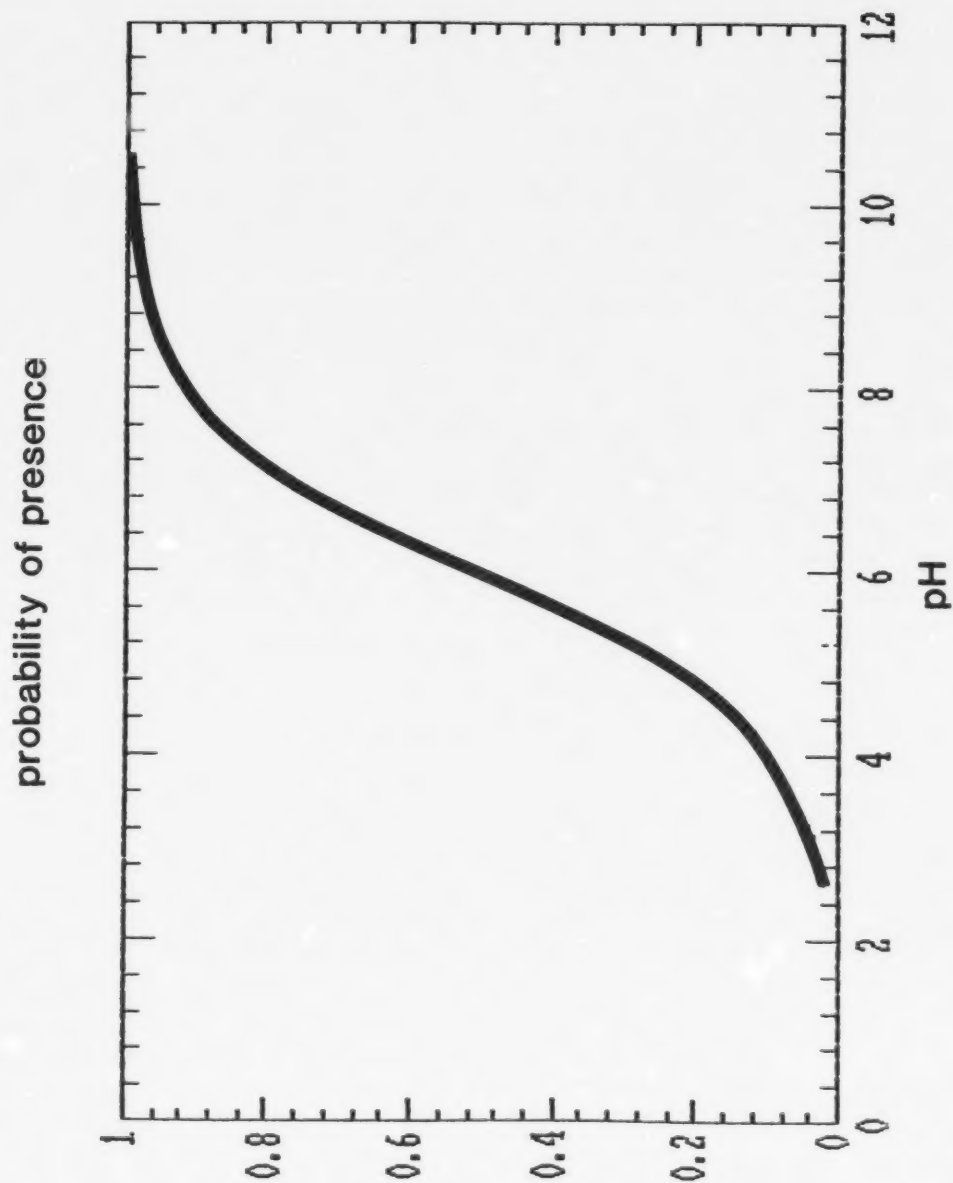


Figure 5.6: Probability of brook trout presence in Adirondack lakes as a function of mean lake pH. Source: Reckhow et al. (1985).

data from the field, have recently been developed to predict the probability of brook trout presence in the Adirondack lakes (Figure 5.6, from Reckhow et al. 1985). This type of probabilistic relationship is extremely useful for assessing the potential for impacts on fisheries exposed to under different deposition regimes.

From the point of view of western Canada, while recognizing that other species may also be affected, one is most interested in species which are economically important and occur on terrain considered to be highly sensitive to acidic deposition. Table 5.6 lists those fish species which are important in western Canada either for commercial, sport, or subsistence fishing. As of 1963 (National Atlas of Canada 1974), the most important inland commercial fisheries in western Canada were lake whitefish (average annual landed value of \$2.7 million), walleye (\$1.9 million), sauger (0.9 million), lake trout (\$0.5 million), and lake herring (\$0.3 million). The western provinces and Northwest Territories were at this time collectively responsible for more than half of Canada's overall landed value of inland commercial fisheries (Manitoba - 30%, Saskatchewan - 10%, Alberta - 7%, and Northwest Territories - 6%). Pacific salmon represents the most economically important fishery, with annual commercial landings of \$150 million and sport fishing values estimated at \$30 million, as of 1981 (Pearse 1982).

At least part of the range of all these species overlaps terrain containing surface waters considered highly sensitive to acidification. Note that walleye, a fish of major importance to both commercial, sport, and subsistence fisheries in western Canada (Scott and Crossman 1973), is one

of the most sensitive species, showing declines in reproduction at pH 6.0 - 5.5 (Figure 5.5). Lake trout, also of major importance in western Canada, is also relatively sensitive, showing declines in reproduction over the pH range 5.6 - 5.4 (Figure 5.5). Pacific salmon, which spawn in many relatively sensitive coastal streams, have not been extensively studied with respect to acidification. Laboratory experiments by Geen et al. (1985) showed high mortality of chinook salmon alevins at pH 5.0. Geen et al. point out that these experiments likely underestimated chinook sensitivity since aluminum concentrations were not elevated (as would likely occur at this pH in the field).

The sensitivity of rainbow trout is also of concern, due to its major importance as a recreational fish. In Figure 5.5, rainbow trout appear to have no successful reproduction below pH 6.5; however this may be an artifact of the scope of existing studies. All seven studies on rainbow trout examined by Baker (1984) were restricted to examining adult mortality or growth. More recent laboratory and field studies of sac fry and fry mortality indicate consistently high mortality at pH 4.5 (75-100%) and variable mortality rates at pH 5.5 (0-90%) (Baker, in Marmorek 1986).

In conclusion, a surface water pH of 5.8 - 6.0 appears to be a reasonable threshold for the protection of most aquatic biota. At this pH range, aluminum concentrations are at a minimum, and thus, not significant (Driscoll and Bisogni 1984), and fish should not be adversely affected. A pH of 5.8 - 6.0 corresponds to an alkalinity of about $10-40 \mu\text{eq}\cdot\text{L}^{-1}$ (the wide range is due to the dependency of pH on both alkalinity and dissolved organic carbon [DOC]). The biota of

Table 5.6

Fish species of direct importance to man in western Canada.

(Species are listed by taxonomic group, not order of importance.
Forage fishes not included.)

Use codes: R = recreational, C = commercial S = subsistence.
[Source: Scott and Crossman 1973])

NAME	MAJOR REGIONS FOUND	USE
1. Pink salmon <u>Oncorhynchus gorbusha</u>	Coastal B.C. and Yukon	R,C,S
2. Chum salmon <u>O. keta</u>	Coastal B.C. and Yukon	R,C,S
3. Coho salmon <u>O. kisutch</u>	Coastal B.C. and Yukon	R,C,S
4. Sockey/Kokanee Salmon <u>O. nerka</u>	Coastal B.C. and Yukon	R,C,S
5. Chinook salmon <u>O. tshawytscha</u>	Coastal B.C. and Yukon	R,C,S
6. Cutthroat trout <u>Salmo clarki</u>	B.C. and S.W. Alberta	R
7. Rainbow trout <u>S. gairdneri</u>	B.C., S. Alberta, S. Saskatchewan, and S. Manitoba	R
8. Brown trout <u>S. trutta</u>	S. Alberta	R
9. Arctic char <u>Salvelinus alpinus</u>	N. N.W.T., and N. Yukon	R,C,S
10. Lake trout <u>S. namaycush</u>	B.C., Alberta, N. Saskatchewan, Manitoba, N.W.T., and Yukon	R,C,S
11. Lake herring <u>Coregonus artedii</u>	Alberta, Saskatchewan, Manitoba, and E. N.W.T.	C,S
12. Arctic cisco <u>C. autumnalis</u>	N. Yukon, and N. N.W.T.	C,S
13. Lake whitefish <u>C. zenithicus</u>	all regions, particularly important in Manitoba, Saskatchewan, and N.W.T.	C,S

Table 5.6 (cont'd)

NAME	MAJOR REGIONS FOUND	USE
14. Broad whitefish <u>C. nasus</u>	Yukon, and N.W.T.	S
15. Arctic grayling <u>Thymallus arcticus</u>	N. provinces, Yukon, and N.W.T.	S,R
16. Eulachon <u>Thaleichthys pacificus</u>	Coastal B.C.	C,S
17. Goldeye <u>Hiodon alosoides</u>	Alberta, Saskatchewan S. Manitoba, and S. N.W.T.	C,R
18. Northern pike <u>Esox lucius</u>	NE B.C., all other regions	C,R,S
19. Yellow perch <u>Perca flavescans</u>	Alberta, Saskatchewan, and Manitoba	C,R,S
20. Sauger <u>Stizostedion canadense</u>	S. Alberta, S. Saskatchewan, and S. Manitoba	C
21. Walleye <u>S. vitreum</u>	Alberta, Saskatchewan, Manitoba, and N.W.T.	C,R,S
22. Freshwater Drum <u>Aplodinotus grunniens</u>	Manitoba	C,R

coloured (high DOC) waters are exposed to more acidic conditions naturally, and are less sensitive to high aluminum concentrations because of complexation of inorganic aluminum by organic ligands. For these waters, a lower threshold may be appropriate. On the other hand, protection of all biota in surface waters (e.g., molluscs) would require a higher pH threshold of about 6.5. The pH range of 5.8 - 6.0 is, however, appropriate for those low alkalinity systems which are most vulnerable to acidification, and probably naturally have pH levels lower than 6.5 at some periods of the year.

To maintain clear surface waters above pH 5.8 - 6.0 year-round (i.e., avoid acidic episodes), the mean annual pH should be somewhat higher. Figure 5.7 shows the frequency distributions of minimum lake pH values for given ranges of mean lake pH (Jones and Cunningham 1985). Lakes with a mean pH of 5.75 - 6.0 commonly have minimum pH values between 4.75 and 5.25. Since the data used to create Figure 5.7 came from stations with a wide range in acidic deposition, it may overestimate the variability of minimum pH within a given subregion. Many of the stations providing data for Figure 5.7 receive much higher levels of deposition than occur in western Canada. Note also that mild pH depressions commonly occur naturally in regions without acidic deposition, due to the dilution effects of meltwater.

5.5.4 Linking Deposition Levels with Water Quality Thresholds

A considerable number of factors control the effects of acidic deposition on surface water quality, including soil/sediment contact, weathering replacement, anion retention,

base cation buffering, and processes in streams and lakes. Much of the current research on aquatic effects of acidic precipitation is concerned with the development and improvement of models to simulate these processes (either explicitly or implicitly) and predict stream and lake water chemistry for given levels of deposition. These models can be roughly categorized into two groups:

- 1) detailed simulation models representing important physical, chemical, and biological processes for a single basin; and
- 2) relatively simple statistical or simulation models which treat the watershed as a "black box", and make predictions on basin or regional scales.

The former class is generally referred to as "process" models and the latter as "empirical" models. These terms, however, should be thought of more as two ends of a spectrum rather than mutually exclusive categories. Even detailed process models rely, at some level, on empirical relationships, and statistical models are usually structured to represent basic processes.

The major trade-off between the two types of models lies in the spatial extent of their applicability. Detailed simulation models generally can only be applied on intensively studied watersheds with two or more years of frequent water chemistry measurements (required to calibrate the model), as well as background data on watershed soils, climate, and forest cover. Empirical models require much less data, and can therefore be applied relatively easily to

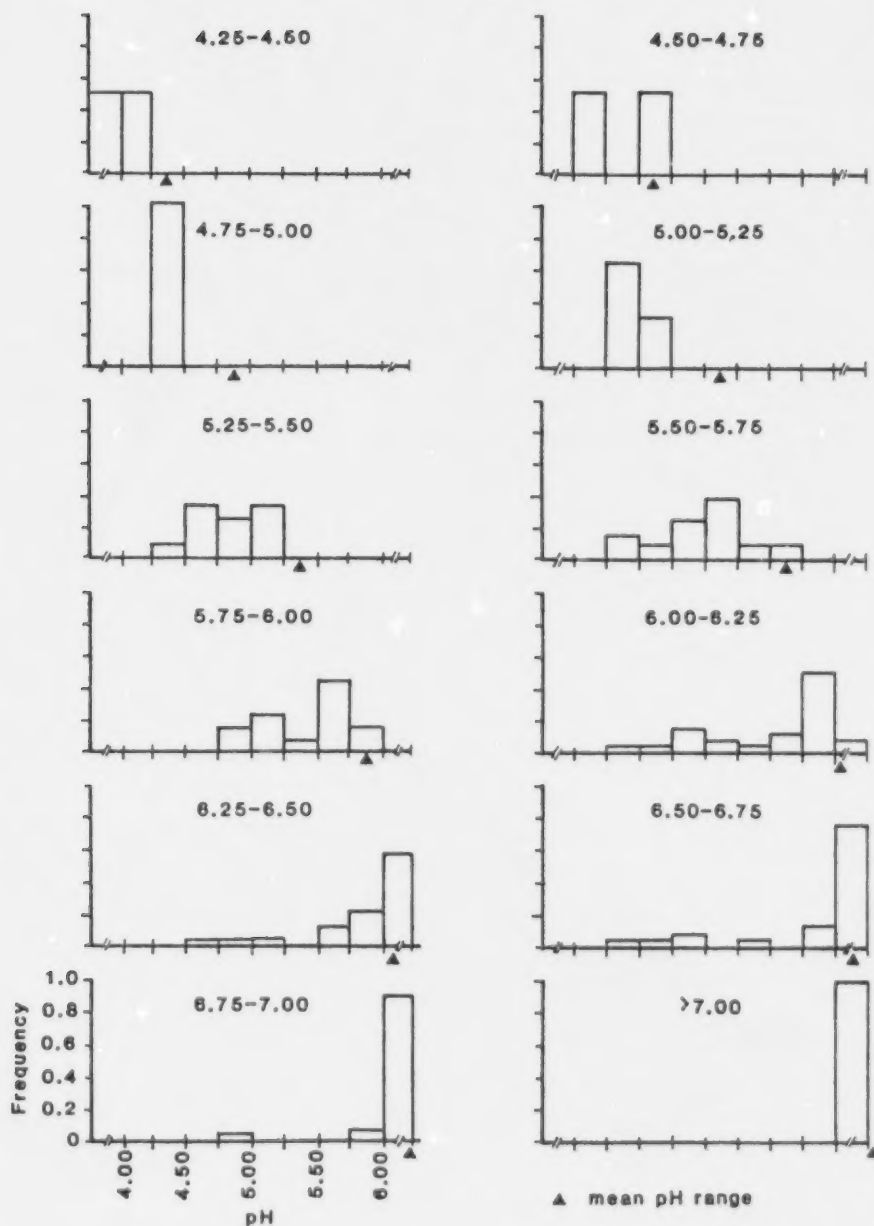


Figure 5.7: Frequency distributions of minimum lake pH values for given ranges of mean pH. Lakes span eight regions of eastern Canada: Experimental Lakes Area, Algoma, Turkey Lakes, Sudbury, Algonquin Park, Muskoka-Haliburton, Forêt Montmorency, and Kejimikujik Park (adapted from Jones and Cunningham 1985).

entire regions, at the cost of making some very simplistic assumptions about aquatic systems.

Successful applications of detailed models can increase understanding of the relative importance of different processes, and, in theory, provide a reasonable estimate of the appropriate target loadings for a few, intensively studied watersheds. Ideally, one would perform model applications on a sufficient number of systems to have a regionally representative subsample. There are, however, very few examples of detailed simulation models being applied to the question of target loadings. The Minnesota Pollution Control Agency (1985) used the Trickle Down model (developed by Schnoor et al. 1982, 1984) to estimate target loadings to four Minnesota lakes. The rain pH required to deplete 30-50% of these lakes' alkalinity (to steady state) ranged from 4.7 to 4.8, but this is considered a conservative estimate since various in-lake neutralization processes (sulphate reduction, cation exchange, and primary production) are not included in the model. The RILWAS project (Regional Integrated Lake-Watershed Acidification Study) applied the ILWAS model (Chen et al. 1983) to a sample of 10-12 lakes in the Adirondacks, but did not specifically address the question of target loadings. In eastern Canada, the Turkey Lakes model (Lam and Bobba 1984; Bobba et al. 1985; Lam et al. 1985) has now been applied to four different regions (Turkey Lakes and Muskoka Haliburton in Ontario, Lac Laflamme in Quebec, and the Mersey River, Nova Scotia), and could potentially address the issue of target loadings.

Empirical models, such as those of Almer et al. (1978), Henriksen (1980), Thompson (1982), and Wright (1983), have

have been the most commonly used method to estimate the level of acidic deposition associated with critical water quality thresholds. These models, and more recent improvements to them (Jones et al. 1984; Nichols and McRoberts 1985), are summarized in Table 5.7. See the MOI (1983) for applications of the first three models shown in Table 5.7 to the question of target loadings. These models are all founded on the principle of charge balance, with the assumption that as surface waters acidify, sulphate replaces bicarbonate. The basic structure of these models consists of three parts:

- 1) an estimate of the system's original alkalinity; that is, the alkalinity it presumably had in the past when strong acid deposition was negligible;
- 2) a representation of the watershed's neutralization of acidic deposition; and
- 3) an estimate of atmospheric acid loading.

Successive improvements to these models is evident from the reduced number of constraints on model applications and predictions in the last column of Table 5.7. A continuing problem is how to model watersheds that retain sulphate without making the models considerably more complex. This may be less of a problem in western Canada, since current evidence (discussed in Section 5.2) indicates there are probably few sulphate-retaining soils in this region. Another unresolved issue is how to represent acidic deposition (wet sulphate, wet plus dry deposition, sulphate plus nitrate, etc.). Nichols and McRoberts have recently cited a new approach (Verry and Harris 1984) to this

Table 5.7

Summary of empirical models which can be used to estimate target loadings. (Ca^*) , (Mg^*) , (Na^*) , (K^*) , (SO_4^*) = current lake concentrations of excess calcium, magnesium, sodium, potassium, and sulphate. $(SO_4^*)_O$ = original concentration of excess SO_4^{2-} ; pH = mean pH of precipitation; $(SO_4^*)_P$ concentration of sulphate in precipitation;

$$F = \frac{\Delta ((Ca^*) + (Mg^*))}{\Delta (SO_4^*)}; T = (\text{terrestrial watershed areas}) /$$

lake area). In Nichols and McRoberts model, $(SO_4^*)_P$ is estimated rather than assumed.

MODEL AND REFERENCE (type of system to which applied)	FACTORS CONSIDERED IN ESTIMATING ORIGINAL ALKALINITY	REPRESENTATION OF WATERSHED NEUTRALIZATION OF ACIDITY	REPRESENTATION OF ACIDIC DEPOSITION	MAJOR CONSTRAINTS (Model Predictions wrong if system has ...)
1. Henriksen nomograph Henriksen 1980 (lakes)	$(Ca^*) + (Mg^*)$	assumed to be zero	regression between (SO_4^*) and pH_P	major S sources/sinks high $(Na^*):(Ca^*)$ high organics not reached steady state watershed neutralization
2. Wright et al. 1983 (lakes)	$(Ca^*) + (Mg^*)$	F	same as Henriksen assumes 30% dry deposition, 1m precipitation	major S sources/sinks high $(Na^*):(Ca^*)$ high organics not reached steady state
3. CDR model, Thompson 1982 (streams)	(Ca^*) , (Mg^*) , (Na^*) , (K)	export of base cations: acid loading	measured wet $(SO_4^*)^*$ runoff = acid load	major S sources/sinks cations export depends on acid loading high organics
4. Jones et al. 1984 (lakes)	$(Ca^*) + (Mg^*)$, F, (SO_4^*) , $(SO_4^*)_O$, T	F, T	measured wet SO_4^{2-} deposition, runoff assumes 25% dry deposition	major S sources/sinks
5. Nichols and McRoberts 1985 (lakes)	(Ca^*) , (Mg^*) , F, (SO_4^*) , $(SO_4^*)_O$	F, $(SO_4^*)_O$	separation of deposition into background and emission related	major S sources/sinks (less of a problem than other models)

issue, which divides wet deposition into background and emission-related categories.

In the more recent models, the original alkalinity is estimated from current base cation concentrations and an "F-factor", which accounts for the rate of change in cation concentrations associated with a given change in sulphate concentrations (presumably as a result of cation exchange processes). An F value of 0 implies no change in cation concentrations (no neutralization except by alkalinity titration), while $F = 1$ implies that all acidic deposition is neutralized. The models then predict the system's steady state alkalinity under a given level of acid loading. The phrase "steady state" refers to the condition of the system when it is at equilibrium with the given level of acid loading. Unlike the detailed simulation models described above, these empirical models do not predict how long it will take the system to reach the predicted condition. This approach avoids the problem of deciding how the F-factor changes with time, but still leaves the difficult question of how to estimate values of F. (For a discussion of this problem see MOI 1983, and Henriksen 1984.)

In spite of these difficulties, empirical models have two major advantages over detailed simulation models:

- 1) they can be relatively easily applied to an extensive region; and
- 2) the effect on predictions of uncertainty in watershed neutralization can be assessed relatively easily.

The first point means that the variability in terrain and climatic factors between regions can be implicitly taken into account via intraregional variability in lake alkalinity. The models can then be used to estimate the proportion of a region's lakes or streams which would be affected by a given level of acidic deposition. One of the most graphic methods of estimating this proportion is through the use of cumulative frequency distributions of lake alkalinity. Figure 5.8a (adapted from Jones et al. 1984) shows the cumulative frequency distribution of original alkalinities for 209 lakes in a secondary watershed of Ontario. Also shown on this figure is the "critical" original alkalinity (CR) necessary to keep lake pH above 5.0 under current levels of deposition, assuming 50% of acidic deposition is neutralized ($F = 0.5$). The intersection of the CR line with the cumulative frequency distribution indicates what percentage of lakes are predicted to acidify under current deposition (about 30%). Figure 5.8b shows that if one assumes none of the deposited acids are neutralized ($F = 0$), over 50% of the lakes are predicted to acidify. Lower values of F imply both a higher original alkalinity and a higher critical alkalinity. Expressing model output in the form of cumulative frequency distributions allows one to relatively easily estimate the predicted effect of different possible target loading regulations, and the approximate range of prediction uncertainty.

5.5.5 Summary

Two basic approaches have been used in developing dose-response relationships for aquatic systems:

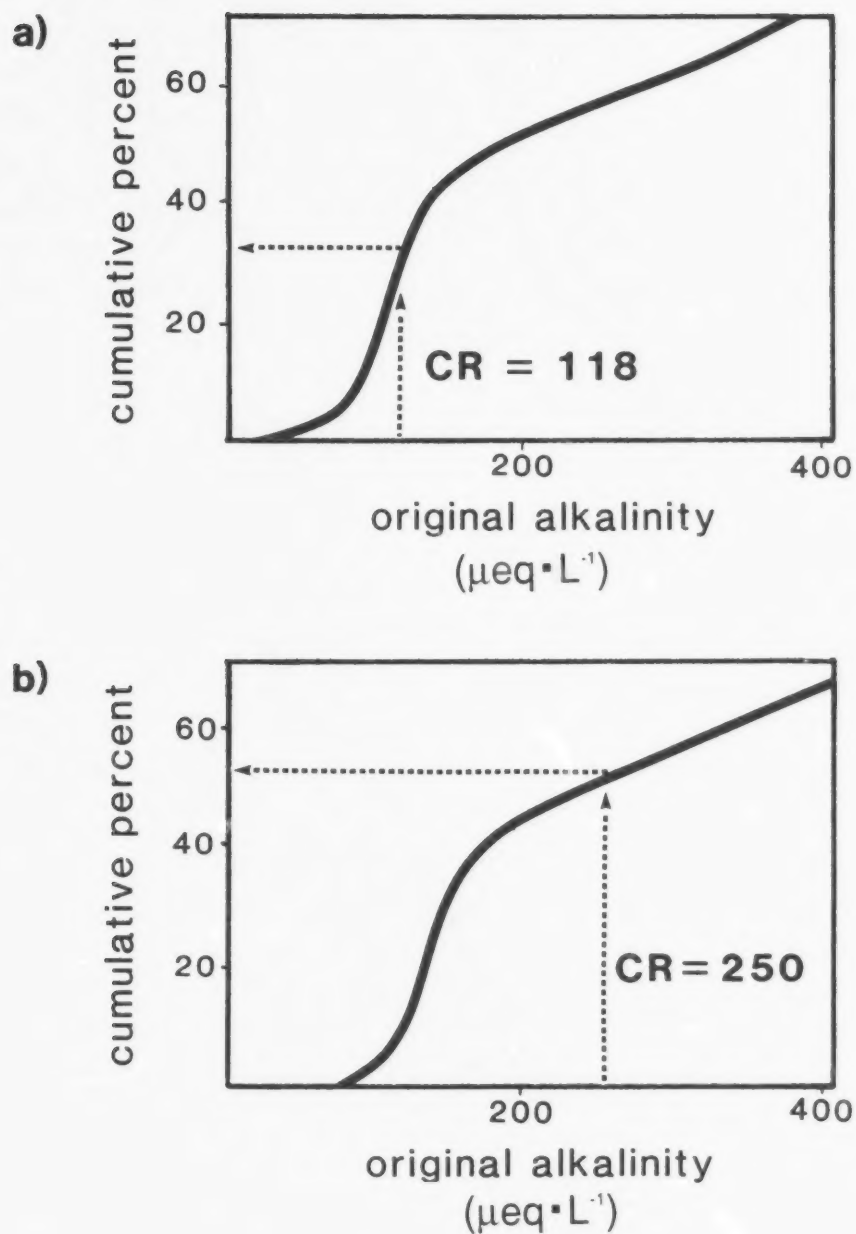


Figure 5.8: Cumulative frequency distribution of the estimated original alkalinities of 209 lakes in a secondary watershed of Ontario, together with the critical original alkalinity (CR) necessary to maintain lake pH above 5.0; a) assuming $F = 0.5$ (50% neutralization); b) assuming from Jones et al. 1984.

- 1) interregional comparisons of the state of health of aquatic biota, or surface water chemistry concentrations, with the level of deposition; and
- 2) disaggregation of the system into two components (deposition ---> surface water responses; surface water quality ----> biological response) with functional relationships developed for each component.

Studies using either approach generally define the dose to aquatic systems as annual wet deposition of sulphate. Though it is recognized that both dry deposition and episodic events are extremely important, there are few data available to meaningfully quantify these contributions. Empirical comparisons indicate damage to aquatic systems at wet deposition thresholds between 10 and 20 $\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ of sulphate. Extrapolation of these comparisons to parts of western Canada may be appropriate, but would not be appropriate in other regions because of large differences in deposition composition, climate, hydrology, soils, and aquatic biota.

Dose-response information indicates that significant biological changes occur in aquatic systems when the mean pH reaches 5.8 - 6.0, and this range appears to be a reasonable threshold to use for establishing target loadings. The actual disappearance of fish species does not occur until the mean pH reaches 4.5 - 5.5 (species-dependent), but recruitment failures can occur at pH 5.5 - 6.0 for the more sensitive species. Western Canada has many economically important fish species in regions with surface waters that are classed as being highly sensitive to acidification.

Both detailed "process" models and relatively simple "empirical" models have been used to estimate target loadings. The latter type of model is easier to implement in regions with a reasonable amount of survey data, such as western Canada. Application of these models must, however, incorporate significant uncertainties in deposition levels, deposition composition, and rates of watershed neutralization of acidic inputs.

5.6 Wetlands

5.6.1 Introduction

While considerable attention has been devoted to examining effects of land use changes on wetlands in western Canada (Lynch-Stewart 1983), relatively little effort has been devoted to researching the effects that atmospheric pollutants have on these systems (Gorham et al. 1984).

Wetlands include all lands that are wet for long enough periods of time to allow development of vegetation (and associated processes) that is distinct from surrounding well-drained areas. Wetlands are generally classified into at least five major categories (Tarnocai 1979): bogs, fens, marshes, swamps, and shallow water areas. The predominance of peat deposits in bogs and fens typically distinguishes them from the other three wetland types; together they are commonly referred to as peatlands. While both bogs and fens are formed on poorly drained soils, bogs are nutrient-poor, receiving nutrient inputs solely from atmospheric sources (ombrotrophic), while fens receive some added nutrients from adjacent upland soils via seepage waters (minerotrophic).

Sphagnum mosses predominate in bogs, whereas sedges and grasses predominate in fens.

Due to the unusual chemistry of peatlands, their importance in global carbon, nitrogen, and sulphur cycles (Gorham et al. 1984), and because they probably serve as major "sinks" (at least temporarily) for atmospherically deposited mineral acids and other pollutants, the following discussion focuses primarily on peatlands.

5.6.2 Effect of Atmospheric Inputs on Peatlands

Given the nature of peatlands (as described in Section 4.2.5), it is widely accepted that atmospheric inputs greatly influence their chemistry. A more important issue to consider here is how increased levels of atmospheric depositions affect the functioning and, ultimately, the structure of peatlands. The key questions, then, are (1) how sensitive are ombrotrophic and minerotrophic peatlands to acidic precipitation; and (2) can a dose-response relationship be established using the data currently available?

Even at sites where acidic deposition is low, ombrotrophic bogs are typically highly acidic (Gorham et al. 1984). This is because in the absence of minerotrophic alkalinity inputs, internal organic acid production overwhelms internal neutralization processes such as sulphur reduction and nitrate assimilation. In such systems, weak acid buffering helps prevent pH changes due to atmospherically deposited mineral acids. Depending on the contribution of nutrients and the alkalinity of incoming seepage waters, fens can be either nutrient-rich or nutrient-

poor. Rich fens usually have relatively high pH values and sufficient alkalinity to buffer pH changes. In contrast, poor fens tend to have intermediate pH values, with little or no bicarbonate alkalinity. Thus, poor fens are hypothesized to be the group of peatlands most sensitive to acidic deposition. Gorham et al. (1984) also found evidence suggesting that even some acidophilous *Sphagnum* species may be susceptible to atmospheric pollutant deposition. They suggest studies of acidification responses should focus on plants and animals with narrow pH ranges that could serve as "indicator species" or make up "indicator communities".

Our current understanding of the processes whereby peatlands retain incoming sulphate, nitrate, ammonium, and metals has been carefully reviewed by Jones et al. (1986). These authors conclude that despite substantial advances in our knowledge of this topic, our understanding is still primarily descriptive rather than predictive. Nevertheless, their review of the evidence to date suggests:

- 1) through biological assimilation and chemical reduction of sulphate, peatlands neutralize incoming acidity and retain sulphate, although it may be flushed periodically from these systems;
- 2) peatlands appear to retain nearly all (>90%) of the atmospherically deposited nitrate and ammonium, with the net acidifying effect depending on relative inputs of each species (nitrate uptake consumes acidity, while ammonium assimilation produces acidity); and

- 3) currently, the evidence as to whether peatlands retain aluminum and manganese is conflicting, although there is general agreement that peatlands act as a source of iron to other surface waters.

It is important for the reader to realize that even when acidic precipitation does not cause harmful impacts to wetland ecosystems, the acidity which travels downstream from such areas is potentially harmful to biotic systems further afield.

5.6.3 Target Loading Considerations

Urban (1984; cited in Minnesota Pollution Control Agency 1985) estimated manmade sulphur loadings of 40 - 48 $\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ would be required to overwhelm the natural buffering capacity of most bogs in the eastern United States. If peatlands in western and northern Canada have similar buffering capacity, they are probably relatively insensitive to atmospheric sulphur loadings. Because this extremely high deposition rate would rarely be experienced in western Canada, and target loadings designed to protect the more sensitive surface waters would probably be roughly one-half or less of this suggested tolerance limit of wetlands, it is reasonable to argue that target loadings for surface waters would be adequate to protect peatlands as well. As pointed out by Gorham (University of Minnesota, pers. comm.), however, since Urban's work deals with sulphate reduction below the water table, it is unclear whether wetland vegetation above the water table would be capable of tolerating such elevated loadings. Consequently, in view of our inadequate understanding of key functional processes in peatlands, we are unable to make substantive predictions at this time.

6.0 APPLICATION OF THE TARGET LOADING CONCEPT TO WESTERN CANADA

As discussed in Section 1, the concept of a target loading is based on the premise that one can identify a threshold level of pollutant deposition, beyond which unacceptable environmental consequences result. Ideally, prescribed target loadings for atmospheric compounds should be based upon a thorough knowledge of: (1) ecosystem components to be protected; (2) pollutants they may be exposed to; and (3) pertinent quantitative dose-response relationships linking concentrations of various pollutants (individually and in combination) with responses by the more "sensitive" components. Sections 2 to 5 of this report have outlined in detail the state of knowledge of these three components for western Canada.

Regarding the application of the target loading concept (described in Section 1.1), western Canada differs from eastern North America in three important respects. First, there are fewer major point sources, and these are spatially segregated. This is an advantage since cause-effect relationships between particular emission sources and particular receptors can be more easily deduced using both modelling and experimental approaches. Second, the western region contains large areas relatively insensitive to atmospheric pollutants together with more sensitive zones. This contrasts with eastern Canada, where a substantial proportion of the region is considered to be sensitive to acidic precipitation (MOI, 1983). Third, western Canada contains regions that receive considerably more precipitation than eastern Canada, as well as drier regions.

The existence of such high spatial variability in receptor sensitivity in western Canada raises two issues. The first concerns the difficulty of deciding how to determine target loadings within a spatially heterogeneous region. The second issue is how to define the boundaries over which regulations would apply. The approach taken in Minnesota (Minnesota Pollution Control Agency 1985), was to define a target loading based on dose-response relationships for surface waters, and apply this only to regions with sensitive surface waters. This may be the most practical way to deal with these two issues, given our current understanding. For much of western Canada (coastal British Columbia, northern Saskatchewan, northwestern and southeastern Manitoba), the regions of high soil sensitivity for forests and crops (Figure 4.2) also contain highly sensitive aquatic systems (Figure 4.4). The Minnesota approach, therefore, seems appropriate in these regions, although the actual target loading levels required for protection may differ from those used in Minnesota. There are, however, three major regions which are classified as highly sensitive on the soils map (Figure 4.2), but only of low to moderate aquatic sensitivity (Figure 4.4):

- 1) a large area in northwestern British Columbia, part of the boreal cordillera ecozone;
- 2) a band of highly sensitive soils running through the mountain ranges of southeastern British Columbia (the Cariboo, Monashee, Selkirk, and Purcell ranges); and
- 3) a large area in northern Alberta between latitude 55 and 59, classified in the boreal plain ecozone.

Additionally, the lower Fraser River valley falls into this

category and although it is not a major region in terms of area, it is a highly important agricultural area.

For these regions, there are essentially two policy options. The first option is to assume the same target loading level as for regions with surface waters of high sensitivity, even though mapping indicates only low to moderate surface water sensitivity. This policy makes the assumption that highly sensitive soils are not likely to be more sensitive than highly sensitive surface waters (essentially the same assumption made in Minnesota). The second policy option is to develop quantitative dose-response relationships for regional soils and forests, and thereby establish scientifically defensible target loadings. The first policy can be developed relatively quickly, but may be difficult to defend. It may be worthwhile, in the short term, to implement ecozone-specific ambient concentration limits to protect terrestrial ecosystems from point source fumigation. Such limits could be established as regulations under existing legislation.

How should target loadings be determined for regions of western Canada containing sensitive surface waters? Ideally, there should be two steps:

- 1) choose a method (or methods) by which one can estimate the loading that maintains lakes or streams of a given sensitivity (e.g. alkalinity, cation export) above a specified threshold; and
- 2) apply the method to existing databases in order to determine what percent of the resource (number or

area of lakes, length of streams) would be protected by various critical loading levels, and what percent would be at risk of acidifying.

To complete the first step, a mix of empirical and mechanistic approaches (as summarized in Tables 1.2 and 5.7) is available. These methods can be currently applied (with appropriate estimates of uncertainty and improved estimates of the "effective deposition") to "typical lake" examples in different regions of western Canada. To estimate the resource at risk, several data gaps need to be filled; these were discussed in Section 4 and are summarized in Section 7. The remainder of this section explores the first step.

Interregional differences in climate, the composition of atmospheric deposition, and watershed neutralization of acidity should be carefully considered when applying existing steady state, empirical models. There are basically two approaches taken by current models. The first approach is to keep acid loading low enough that watershed generation of bicarbonate is maintained. The basic method (outlined in Henriksen et al. 1986) is to obtain a rough estimate of the weathering rate from base cation export, and keep deposition less than this weathering rate. Base cation export is proportional to a lake's mean concentration of base cations and the volume of lake outflow. Thus, two watersheds with the same lake chemistry but different amounts of precipitation will have different computed target loadings (proportional to the level of precipitation).

The second method involves specifying a threshold lake alkalinity which must be maintained, and back-calculating

what deposition will produce it given the lake's current alkalinity. This approach is most easily explained by first showing what the steady state alkalinity will be under a given loading. Existing empirical models are essentially of the form:

$$\text{Alk}_T = \text{Alk}_O - \frac{\text{TL} * (1-F)}{R} \quad (6.1)$$

where:

Alk_T = the alkalinity (in $\mu\text{eq}\cdot\text{L}^{-1}$) the system will eventually reach under the target loading;

Alk_O = the system's original alkalinity ($\mu\text{eq}\cdot\text{L}^{-1}$);

TL = a given target loading or assumed level of deposition (in μeq of hydrogen ions $\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$);

F = fraction of annual deposition neutralized by watershed or in-stream/in-lake processes; and

R = runoff ($\text{m}\cdot\text{yr}^{-1}$).

The definition of F, given above, is operationally correct, but slightly different from that generally shown (as in Henriksen 1984). Strictly speaking, F equals the change in lake [Ca + Mg] for a given change in lake sulphate concentrations, as described in Table 5.7. The way in which F is used in empirical models, however, makes it equivalent to the definition used here. Thus, though current models do not have explicit terms for in-lake alkalinity generation, which can be very important in some lakes (Schindler et al. 1986; Cook et al. 1986), they can incorporate this process implicitly by increasing the level of F.

Equation 6.1 can be rearranged to estimate a target loading which will prevent the system's alkalinity from declining below Alk_T :

$$\text{TL} = \frac{(\text{Alk}_O - \text{Alk}_T) * R}{(1 - F)} \quad (6.2)$$

Illustrative calculations using Equation 6.2 are shown in Table 6.1 for four regions mapped as highly sensitive (Figure 4.4) but with different amounts of runoff. The runoff values assigned are the midpoints of the ranges shown on a runoff map in the Hydrological Atlas of Canada (1978). The Table estimates the target loadings which would be sufficient to reduce the alkalinity of two different lakes (one with an original alkalinity of $50 \mu\text{eq}\cdot\text{L}^{-1}$, the other with $100 \mu\text{eq}\cdot\text{L}^{-1}$) to $32 \mu\text{eq}\cdot\text{L}^{-1}$, corresponding to a pH of 5.8 in relatively clear surface waters. (If a pH of 6.5 were used, then the target loading would be essentially zero, since lakes of pH 6.5 have an alkalinity of approximately $50 \mu\text{eq}\cdot\text{L}^{-1}$.)

Three different assumptions are made for watershed neutralization ($F = 0$, $F = 0.4$, and $F = 0.8$), and the computed target loadings are expressed both in units of hydrogen ion loading and sulphate loading ($\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$), assuming a 1:1 ratio of hydrogen ion to sulphate equivalents.

Note that this assumption will underestimate the correct target loading in prairie regions, since alkaline particulates produce a lower ratio of hydrogen ions to sulphate equivalents (as explained in Section 2). The sulphate target loading shown in Table 6.1 can be considered as "effective acid deposition expressed in terms of sulphate"; however, to be strictly correct, it would be

Table 6.1

Illustrative calculations of target loadings for surface waters of varying sensitivity for different areas of western Canada, using the formula:

$$TL = \frac{(Alk_O - Alk_T) * R}{(1 - F)}$$

where, TL = target loading in $\mu\text{eq}\cdot\text{L}^{-1}$ [as $\text{kg SO}_4^{2-}\cdot\text{ha}^{-1}$]**;

Alk_O = original alkalinity ($\mu\text{eq}\cdot\text{L}^{-1}$);

Alk_T = target alkalinity ($\mu\text{eq}\cdot\text{L}^{-1}$) for protection (here assumed equal to 32 $\mu\text{eq}\cdot\text{L}^{-1}$);

R = runoff ($\text{m}\cdot\text{yr}^{-1}$); and

F = fraction of annual deposition neutralized by watershed or in-stream/in-lake processes.

See text for derivation.

LOCATION	ASSUMED RUNOFF ($\text{m}\cdot\text{yr}^{-1}$)	ESTIMATED TARGET LOADING ($\mu\text{eq}\cdot\text{m}^{-2}$)($\text{kg SO}_4^{2-}\cdot\text{ha}^{-1}$)					
		Alk_O					
		50 $\mu\text{eq}\cdot\text{L}^{-1}$			100 $\mu\text{eq}\cdot\text{L}^{-1}$		
		F=0	F=0.4	F=0.8	F=0	F=0.4	F=0.8
1. NE Alberta SE Manitoba	0.1	1.8 [0.9]	3.0 [1.4]	9.0 [4.3]	6.8 [3.3]	11.3 [5.4]	34.0 [16.3]
2. N Saskatchewan NW Manitoba	0.15	2.7 [1.3]	4.5 [2.2]	13.5 [6.5]	10.2 [4.9]	17.0 [8.2]	51.0 [24.5]
3. Yukon	0.3	5.4 [2.6]	9.0 [4.3]	27.0 [13.0]	20.4 [9.8]	34.0 [16.3]	102.0 [49.0]
4. Coastal B.C.	1.5	27.0 [13.0]	45.0 [21.6]	135.0 [64.8]	102.0 [49.0]	170.0 [81.6]	510.0 [244.8]

**effective acid deposition expressed as sulphate deposition (see text)

Note: to be strictly correct, these data would have to be adjusted for ammonium loadings, nitrate loadings, and alkaline dustfall.

necessary to adjust the data for ammonium loadings, nitrate loadings, and alkaline dustfall. More analyses of deposition data are required to determine what level of wet deposition of sulphate this corresponds to within each region of western Canada.

The results in Table 6.1 are thought-provoking. Equation 6.2 assumes that if all other factors are equal, target loadings should be proportional to runoff. The target loadings computed in Table 6.1 suggest that when wet and dry regions are compared, all other factors are probably not equal. For example, Summers (1986) estimated the current rate of sulphate deposition in northern Saskatchewan is about $5 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ (including both wet and dry forms). The second row of Table 6.1 indicates that this loading would be sufficient to acidify (to pH 5.8) a northern Saskatchewan lake with an original alkalinity of $50 \text{ }\mu\text{eq}\cdot\text{L}^{-1}$ and a watershed which neutralized 40% of incoming deposition ($F = 0.4$), or a lake with an original alkalinity of $100 \text{ }\mu\text{eq}\cdot\text{L}^{-1}$ and $F = 0$. By way of comparison, loadings ten times as great (i.e. 21.6 and $49 \text{ kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$, indicated in the bottom row of the Table) would be required to acidify British Columbia lakes with the same original alkalinities and F-factor values as those in Saskatchewan.

Intuitively, these figures appear to overestimate the sensitivity of northern Saskatchewan lakes, and underestimate the sensitivity of British Columbia lakes. One would expect greater soil contact time (and therefore higher F values) in drier climates, since the surface soils are less likely to become saturated; similarly, watersheds in wetter climates should have lower F values. Unfortunately, there is very

little evidence available to assess how F does vary with runoff. Interregional comparisons of long-term monitoring studies (with careful input-output budgets and experimental manipulations) would be required to quantify this relationship. For now, the professional judgement of experts regarding the probable range of F for different regions could be used to obtain interim target loading estimates (or ranges).

The MOI (1983; pg. 3-211) estimated that annual wet sulphate target loadings of 3 kg per hectare would keep a lake with an initial calcium concentration of $50 \mu\text{eq}\cdot\text{L}^{-1}$ ($45 \mu\text{eq}\cdot\text{L}^{-1}$ alkalinity) at pH 5.8, assuming $F = 0$ and runoff = $0.3 \text{ m}\cdot\text{yr}^{-1}$. This is essentially the same as the cell of Table 6.1 for the Yukon, with $\text{Alk}_0 = 50 \mu\text{eq}\cdot\text{L}^{-1}$ and $F = 0$. Thus, the two methods are equivalent.

Existing data suggest that northern Saskatchewan lakes are more tolerant than is implied by Table 6.1. A survey of 125 Shield lakes in northern Saskatchewan (Liaw and Atton 1981) found no lakes with alkalinities less than $100 \mu\text{eq}\cdot\text{L}^{-1}$, and a mean alkalinity of $560 \mu\text{eq}\cdot\text{L}^{-1}$. Though Liaw and Atton only used available data, and not a random sample of lakes, their data suggest that original alkalinities (and probably F values) were sufficiently high to have prevented acidification of northern Saskatchewan lakes under current loadings.

Though Table 6.1 may raise more questions than it answers, it does provide a "straw-man" hypothesis from which to consider possible target loading levels in western Canadian regions with sensitive waters. The principle of different target loading standards for climatically distinct

regions appears reasonable; the important question is, How different? Consultation with a group of experts could certainly narrow the range of target loading estimates shown in Table 6.1.

Summary

Knowledge of three elements is necessary for prescribing target loadings: (1) the ecosystem components to be protected; (2) the pollutants to which they may be exposed; and (3) pertinent quantitative dose-response relationships. Existing studies have only suggested target loadings for aquatic systems, as quantitative dose-response relationships are not available for soils and forests. Previous studies have recommended target loadings for aquatic systems in the range from 10 - 20 kg of sulphate·ha⁻¹·yr⁻¹ as wet deposition, or a mean rain pH from 4.3 to 4.8.

In western Canada, the high spatial variability in receptor sensitivity raises the issues of which ecosystem components to use to determine target loadings, and how to define the boundaries within which the targets will be applied.

For regions with highly sensitive soils and aquatic systems, the logical approach is to develop target loadings for surface waters and assume these standards will protect soils, forests, and crops. Existing models can be applied to help define these targets, but must be adapted to reflect properly, regional differences in deposition composition, runoff, and watershed neutralization.

In regions with high soil sensitivity, but moderate to low aquatic sensitivity, quantitative dose-response relationships for soils, wetlands, and forests must be developed before scientifically defensible target loading levels can be determined.

Until this has been done, limits to the rate of acid deposition could be based on ambient concentrations or target loadings. Ambient concentrations could be based on the sensitivity of specific ecozones to damage by fumigation. Alternately, target loadings could be based on the rates of deposition found to be acceptable for highly sensitive surface waters.

7.0 RECOMMENDATIONS

7.1 Overview

This chapter focuses primarily on:

- 1) suggested steps leading to development of interim target loadings (TL) for western Canada;
- 2) major sources of scientific uncertainty and significant gaps in the current databases that represent barriers to development and adoption of defensible target loadings; and
- 3) corrective actions recommended to reduce the scientific uncertainty and improve the databases.

The following section (Section 7.2) deals with the first item, while the second and third items are dealt with in Section 7.3.

7.2 Development of Interim Target Loadings

To develop interim target loadings designed to protect western Canadian ecosystems from adverse effects due to acidic deposition, there are two major requirements, both of which must be satisfied:

- 1) availability of methods for objectively deriving target loadings; and
- 2) the data necessary to complete calculations underlying scientifically defensible target loadings.

Readers familiar with the information presented throughout this report will recognize that both the methods and availability of data vary according to receptor, and that of all the receptors identified and discussed in Chapter 5, aquatic systems are the ones for which this information is most complete. Hence, it will be possible to develop target loadings for this receptor prior to establishing targets for other receptors, such as forests or wetlands.

To expedite orderly development and adoption of target loadings for all sensitive ecosystem components, we recommend that two complimentary strategies be adopted simultaneously. They are:

- 1) identify and pursue those steps that lead most directly and immediately to development of interim target loadings for sensitive aquatic systems; and
- 2) initiate needed research and actions that will be required for refinement of aquatic target loadings over the longer term and will permit development of targets for other receptors, such as forests and wetlands.

These two strategies are detailed in the following sections.

7.2.1 Steps Leading to Interim Target Loadings

In order to develop interim target loadings within one year, it will be necessary to deal with major sources of uncertainty in the areas of deposition loadings, soils, and

Table 7.1

Development of interim target loadings
(TL = target loadings)

<u>CATEGORY</u>	<u>MAJOR UNCERTAINTIES</u>	<u>CORRECTIVE ACTIONS</u>
Loadings	• effective load • dry deposition • interregional variability	• correlate wet [SO_4^-] and [Ca^+] • estimate by modelling • analyze different stations
Soils	• TL to maintain cations • weathering rates	• use simple models • input/output analyses
Forests/Wetlands	• range of TL	• accept aquatic TL • design dose-response experiments
Surface Waters	• range of TL • resource at risk	• best guesses for parameters • apply existing models • frequency distribution of alkalinity • assess dataset biases, counts and measures information
Target Loadings	• is aquatic TL adequate?	• compare aquatic TL with soil TL

surface waters. It is unrealistic to expect that uncertainties associated with forest effects can be adequately dealt with in this time period. Likewise, it would be difficult to incorporate agriculture into the interim target loadings due to the diversity of agricultural crops and the lack of dose-response data for most western Canadian crop species. As well, most of the intensively farmed prairie soils are relatively well-buffered and thus insensitive to acidification. Table 7.1 presents a summary of the major uncertainties and corrective actions necessary to establish interim target loadings.

The three major uncertainties associated with loadings are: (1) effective acid loadings (acid loadings minus alkaline particulate loadings); (2) dry deposition; and (3) interregional variability in sulphate-hydrogen ion ratios. Resolution of these uncertainties is necessary to identify areas where current loadings exceed whatever interim target loadings are established. Fortunately, each of these uncertainties can be addressed relatively quickly using the corrective actions shown in Table 7.1.

In the area of soils, by estimating weathering rates it may be possible to use simple models to establish a range of target loadings. This is based on the simplifying premise that soil acidification will eventually occur in any areas where atmospheric hydrogen ion loadings exceed the rate of cation release by weathering reactions. Weathering rates are difficult to measure, but can be estimated for areas with calibrated watersheds using input/output data. Since there are already a few such intensively studied, calibrated watersheds in western Canada (see Section 5.2.2), existing simulation models can be readily applied to datasets from

these sites. Such an exercise will also help provide a better understanding of regional differences in weathering rates.

There are two major uncertainties with respect to establishing target loadings for aquatic ecosystems: (1) the appropriate range of target loadings; and (2) a quantitative indication of the resources at risk. As indicated in Table 7.1, one of the best ways to evaluate the aquatic resources at risk is to examine the frequency distribution of surface waters with a given alkalinity. Unfortunately, as indicated in Section 4.2.3, there are considerable biases associated with water quality datasets in some provinces; consequently, only a very preliminary evaluation of resources at risk is possible in some western provinces. In determining target loadings for aquatic systems, we suggest that the most likely range of F values be selected by soliciting the opinions of other experts to compare with those we have presented in Table 6.1. Existing models should then be used to help determine the acceptable range of critical loads (see Section 1).

A reasonable question to ask is whether target loadings designed to protect sensitive aquatic systems will also provide adequate protection to other ecosystems. One early check that can be done is to compare target loadings determined for aquatic systems with those for soils. Admittedly, this will provide little insight into whether aquatic target loading will, for instance, provide adequate protection to forest vegetation, but it will lend added support to those portions of the ranges that may overlap.

For regions containing highly sensitive soils but aquatic systems with only low to moderate sensitivity (as discussed in Section 6), ecozone-specific ambient concentration limits could be implemented, on the short term, to protect terrestrial ecosystems from point-source fumigation. Such limits could be established as regulations under existing legislation if current standards would be inadequate to protect certain ecosystems.

Another alternative worth exploring is to consider developing several target loading values which would then be applied to geographic regions having a specific combination of sensitive systems. For example, one target loading might apply to all areas having both sensitive surface waters and sensitive soils, while another might be applied to regions having only moderately sensitive soils and moderately sensitive surface waters.

7.2.2 Initiation of Needed Research

At the same time that interim target loadings are being developed, research needed to help clarify longer term uncertainties should be initiated. We recommend especially that dose-response experiments in forested ecosystems be started as soon as possible, since these systems are slow to show a response. It is also important to ensure continued support for ongoing calibrated watershed studies.

Table 7.2

Refinement of interim target loadings
(TL = target loadings)

<u>CATEGORY</u>	<u>MAJOR UNCERTAINTIES</u>	<u>CORRECTIVE ACTIONS</u>
Loadings	• dry deposition • monitoring network gaps • imports/exports • effective load	• install equipment • evaluate station location • modelling • watershed NH_4^+, NO_3^- uptake
Soils	• soil acidification from fertilizers • dose-response	• examine soil scientific literature • experiments
Forests/Wetlands	• dose-response • wetland sensitivity • number and location of bogs	• experiments • experiments • counts and measures
Agriculture	• Selenium:Sulphur (cattle and wildlife)	• examine literature
Surface Waters	• dose-response western Canada species • in-lake alkalinity generation • resource at risk	• experiments • examine literature • develop empirical models • resurvey some areas
Target Loadings	• do aquatic TL protect other ecosystems?	• compare TL values across all categories

7.3 Refinement of Target Loadings

Refinement of aquatic target loadings over the longer term and development of target loadings for other receptors, such as forests and wetlands, will require that the major uncertainties listed in Table 7.2 be addressed using the corrective actions listed in this table; highlights are presented in the following sections.

Loadings

The four key sources of uncertainty are: (1) dry deposition quantities and geographic patterns; (2) gaps in the monitoring networks; (3) imports and exports of pollutants; and (4) effective acid loadings. In order to overcome items 1 and 2 (and perhaps 3), it will be necessary to carefully evaluate the current monitoring network, both in terms of geographic coverage as well as the measurements taken at each site. Pollutant imports and exports can also be examined through modelling efforts. Effective acid loadings must take into account more than simply acid loadings minus alkaline particulate loadings; for example, the effects of ammonium and nitrate uptake in the watershed can play an important role in determining the effective acid loading.

Soils

To refine the interim target loadings for soils, it will be necessary to have dose-response information linking acid inputs with responses by certain, well-defined indicators. Collecting these data will require experimentation; although

most of the major questions can be answered through laboratory experiments, it will be necessary to verify the laboratory data through at least some field experiments.

Forests/Wetlands

Since the current state of understanding regarding effects of acidic deposition on forests and wetlands is inadequate even for development of interim target loadings, it is obviously necessary to devote considerable attention to this topic. In some provinces, such as Alberta, a quantitative evaluation of the resources at risk might show that the risks to forests/wetlands are greater than to surface waters. It is very important to clearly recognize that if uncertainties associated with effects to these ecosystems are to be resolved in a reasonably rapid and efficient manner, then it will be necessary to initiate dose-response experiments in the near future, ideally within the coming year. As an initial starting point, we recommend that studies be undertaken to more carefully evaluate effects associated with large point-sources in western Canada (see Section 3.3). An example of such studies is the work being undertaken by Wotton and coworkers in the Thompson area of northern Manitoba.

The issue of wetland sensitivity can be most effectively addressed through experimental acidification. In the event that sensitive wetland types are identified, it will be necessary to have good data on the number and location of bogs in order to quantify the resource at risk.

Agriculture

There are relatively few major uncertainties associated with agriculture. Perhaps the two most important are: (1) the interactions of selenium and sulphur and the consequent effects on cattle and wildlife of associated disorders such as selenium deficiencies; and (2) effects of acidic deposition on crop species in western Canada. From examining the existing literature, it should be possible to deal with the first item. Dealing with the second will be somewhat more complicated, since the data are simply not available. Thus, if the data were judged to be required, experimentation would be necessary. It is possible, though, that expert opinion would judge such steps to be unnecessary at this time.

Surface Waters

Refinement of aquatic target loadings over the longer term will require that several key uncertainties are addressed. Areas requiring attention include: (1) dose-response data for aquatic organisms found in western Canada; (2) in-lake alkalinity generation; and (3) improved quantitative estimates of the aquatic resources at risk. To resolve issue one, experimentation will be necessary. Attention should be given not only to single species tests, but also to some of the newly emerging methods of ecotoxicology, where multiple species/life stages are exposed under more natural conditions. The question of in-lake alkalinity generation can perhaps be partially resolved by keeping abreast of recent work by leading scientists such as Cook and Schindler. In addition, development of relatively

simple empirical models to describe the generation of alkalinity in lakes may be helpful in answering the question for surface waters in western Canada.

The single most important set of actions required to support refinement of target loadings in western Canada is the need to resurvey some areas. As pointed out in Section 4.2.3, the current water quality databases for some provinces are simply inappropriate and inadequate to quantitatively describe the resource at risk. Without such information it is difficult to persuasively argue for establishment of target loading values, since the "cost" of inaction remains unclear.

We also recommend establishment of calibrated watersheds in two new locations: one in either northern Saskatchewan or northwestern Manitoba, and another in the Yukon. These sites would complement those already established in other parts of western Canada (i.e., ELA [western Ontario], Haney [British Columbia], Marmot Basin [Alberta]), thereby providing a regional network of intensively studied sites. Through comparative studies, it would then be possible to gain a better understanding of the regional differences in both aquatic and terrestrial systems.

Target Loadings

If the adopted strategy is to establish interim aquatic loadings within the first year with an intention to develop interim targets for other receptors, such as forests and wetlands, within perhaps three years, then it will be necessary at some point to ask the question, Do aquatic target loadings afford adequate protection to all major

ecosystems? We suggest that the process of refining aquatic target loadings will not only facilitate development of target loadings for other receptors, since experience and strategies developed during establishment of aquatic target loadings will help in identifying fruitful approaches in other areas, but will also make comparison of target loadings across all categories easier, since later target loading will presumably be similarly structured.

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9.0 GLOSSARY OF ACRONYMS, MODELS, SYMBOLS, AND
SPECIALIZED TERMS

ACRONYMS

ACSCEQ	Associate Committee on Scientific Criteria for Environmental Quality
AIRSHED	See <u>MODELS</u> below
ADOM	See <u>MODELS</u> below
ADRP	Acid Deposition Research Program
AES	Atmospheric Environment Services (Environment Canada)
APN	Air Pollution Monitoring Network (AES)
CANSaP	(Also: CANSAP). Canadian Network for Sampling Precipitation (AES)
CAPMoN	(Also: CAPMON). Canadian Air and Precipitation Monitoring Network (AES)
CEC	Cation Exchange Capacity
CFS	Canadian Forest Service
CMC	CMC grid (unidentifiable acronym)
DFO	Department of Fisheries and Oceans (Environment Canada)
DOC	Dissolved Organic Carbon
ELA	Experimental Lakes Area (Western Ontario)
EPA	Environmental Protection Agency (USA)
EPS	Environmental Protection Service (Environment Canada)
FISHER	See <u>MODELS</u> below
FRNRF	Foundation for Research of Natural Resources in Finland
GVRD	Greater Vancouver Regional District (British Columbia, Canada)
IBP	International Biological Program
LRTAP	Long-Range Transport of Air Pollutants

ACRONYMS continued

MAP35

MESOPUFF See MODELS below

MOI U.S./Canada Memorandum of Intent

MPCA Minnesota Pollution Control Agency

OME-STATMOD See MODELS below

QA/QC Quality Assurance/Quality Control

NADP National Atmospheric Deposition Program (USA)

NAPAP National Acid Precipitation Assessment Program (USA)

NAPS National Air Pollution Surveillance Program (EPS)

NAS National Academy of Sciences (USA)

NEIS National Emission Inventory System

NRC National Research Council (Canada), (See also: NRCC)

NRC National Research Council (USA)

NRCC National Research Council of Canada (See also: NRC)

PAH Polynuclear Aromatic Hydrocarbons

RILWAS Regional Integrated Lake-Watershed Acidification
Study

PLMSTAR See MODELS below

PQMP Precipitation Quality Monitoring Program (Alberta)

RCDM See MODELS below

RTM III See MODELS below

THC Total Hydrocarbons

TL Target Loading

TSP Total Suspended Particulate (airborne)

VOC Volatile Organic Carbon

MODELS

ADOM	Acid Deposition and Oxidant Model
AIRSHED	Airshed
FISHER	Statistical estimates of regional transport and acidic deposition
MAP3S	Multistate Atmospheric Power Production Pollution Study
MESOPUFF	Mesoscale Puff Model
OME-STATMOD	Ontario Ministry of the Environment-Statistical Model
PLMSTAR	a photochemical model (PLM = plume)
RCDM	Regional Climatological Dispersion Model
RTM III	Regional Trajectory Model-III
SERTAD	Modelling system for: Statistical Estimates of Regional Transport and Acidic Deposition (an adaptation of FISHER'S model)

SYMBOLS

Al	Aluminum
Ca	Calcium
Cl ⁻	Chloride ion
Ca ⁺⁺	Calcium (free ionic calcium)
d	day
eq	equivalents
g	gram
h	hour
H ⁺	hydrogen ion (hydronium ion)
HNO ₃	nitric acid
H ₂ S	hydrogen sulphide
h·d ⁻¹	hours per day
ha	hectare (1 hm ² or 10,000 m ²)

SYMBOLS continued

K Potassium

k kilo (1,000 or 10^3)

kg kilogram

km kilometres

ktonnes kilotonnes

kg·ha⁻¹ kilograms per hectare

kg·ha⁻¹·yr⁻¹ kilograms per hectare per year

kg SO₄⁼·ha⁻¹·yr⁻¹ kilograms of sulphate ion per hectare per year

L litre

m metre

m- milli (0.001 or 10^{-3}) e.g. millimole

mo month

mo⁻¹ per month

μ micro (0.000,001 or 10^{-6})

μeq·L⁻¹ microequivalents per litre

μeq·m⁻²·yr⁻¹ microequivalents per metre squared per year

μm microns (micrometres)

μMole micromole

mg milligram

mg·m⁻³ milligrams per cubic metre

Mg⁺⁺ Magnesium ion

Mn Manganese

Na Sodium

NH₄⁺ Ammonium ion (ionized ammonium)

SYMBOLS continued

NO Nitrous oxide (cf. NO_x)

NO_2 Nitrogen dioxide

NO_3^- Nitrate ion

NO_x Nitrous oxides (cf. NO)

O_3 ozone

pH potential of hydrogen--a measure of the acidity or alkalinity of a solution

ppm parts per million

S Sulphur

Se Selenium

SO_2 Sulphur dioxide

$\text{SO}_4^{=}$ Sulphate ion

SO_x Sulphur oxides

t tonne

tonne 1,000 kilograms

$\text{tonnes} \cdot \text{km}^{-2} \cdot \text{yr}^{-1}$ tonnes per kilometre squared per year

wk week

wks weeks

yr year

< less than

> greater than

SPECIALIZED TERMS (in the context of Target Loading)

F-factor An empirical model which accounts for the rate of change in cation concentrations in water, associated with a specified change in sulphate concentrations.

F=0 Implies that there is no change in cation concentrations, i.e. no neutralization except by alkalinity titration.

F=1 Implies that all acidic deposition is neutralized.

Long-range Greater than 500 km. Refers to atmospheric processes, or modelling of such processes, over a range of 500 km or more (cf. mesoscale).

Mesoscale Between 200 and 500 km. Refers to atmospheric processes, or modelling of such processes, over a range of 200-500 km (cf. long-range).

Target Loading (TL) A concept pertaining to permissible rates of acid deposition based on the premise that it is possible to identify the point beyond which unacceptable environmental consequences would result.

Western and Northern
LRTAP Consultative
Committee A committee of deputy ministers of environment ministries in western and northern Canada--British Columbia, Alberta, Saskatchewan, Manitoba, and the Northwest Territories--whose function is to coordinate monitoring, research, and other activities related to acid rain, and the long-range transport of air pollutants.

Western and Northern
LRTAP Technical
Committee A committee of senior officials of environment ministries in western and northern Canada--British Columbia, Alberta, Saskatchewan, Manitoba, and the Northwest Territories--whose function is to design monitoring, research, and other activities related to acid rain, and the long-range transport of air pollutants.

SPECIALIZED TERMS (in the context of Target Loading)
(Continued)

Western and Northern	The term "Western and Northern" replaces the earlier convention of "Western". This convention changed around 1985; however, representation on the committee has not changed.
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APPENDIX I: Major point sources of SO₂ in western Canada

Province	Location	Description	Emissions (tonnes)	Reference
British Columbia	Trail	Smelter	23,700	BC MOE (1985)
	Fort Nelson	Gas Plant	11,000	
	Taylor	Gas Plant	10,200	
	Kimberley	Pulp Mill	8,800	
	Port Mellon	Pulp Mill	8,000	
	Powell River	Pulp Mill	7,700	
	Port Alice	Pulp Mill	7,300	
	Chetwynd	Pulp Mill	5,500	
Alberta	Suncor, Fort McMurray	Oil Sands	129,000	Alberta Environ- ment (1982)
	Syncrude, Fort McMurray	Oil Sands	106,600	
	Sundance	Power Plant	78,800	
	Kaybob South	Gas Plant	72,300	
	Ram River	Gas Plant	64,600	
Saskatchewan	Boundary Dam	Power Plant	28,600	Saskatchewan Environment (1983)
	Poplar River	Power Plant	39,200	
Manitoba	Flin Flon	Smelter	293,000	CEC Order, 1981
	Thompson	Smelter	413,900	CEC Order, 1983

APPENDIX II: Areas of various regions in
in Canada

Region	Area (km ²)
British Columbia	948596
Alberta	661185
Saskatchewan	651900
Manitoba	650087
Northwest Territories	3379683
Eastern Canada	3148364
Western Canada ¹	2911768
Western Canada ²	6291451
Yukon	536324
All Canada ³	9976139

¹ Excludes N.W.T.

² Includes N.W.T.

³ Area excluding N.W.T. is 6060132 km².

APPENDIX III: Reported residence time of H₂S

Residence Time	Remarks	References
2h	Presumed to be due to surface catalyzed reaction with O ₃ .	Cadle and Ledford, 1966
4-35 h (marine) 10-32 h (terrestrial) 7-33 h (global)	Based on Friend's estimates of H ₂ S production and concentrations measured in study.	*Slatt et al. 1978
18 h	Reaction with OH radical (California)	*Sprung 1977
22.3 h (summer) 41.9 d (winter)	Estimated life of H ₂ S due to homogeneous gas-phase reactions at 55°N latitude	*Bottenheim and Strausz 1980
<1 d	Global sulphur budget	*Kellogg et al. 1972
~1 d, ~12 h	Assumed first order reaction	*Jaeschke et al. 1978, 1980
~1 d	OH + H ₂ S → H ₂ O + SH Average OH radical concentration of ~3 x 10 ⁶ molecules·cm ⁻³	*Perry et al. 1976
27 h	Reaction of OH radical and measured H ₂ S concentrations	*Delmas et al. 1980
30 h	Reaction with OH radical at concentration of 2.3 x 10 ⁶ molecules·cm ⁻³	*Stuhl 1974
1.5 d	Reaction with OH radical	*Sze and Ko 1980
1.7 d	Global sulphur budget	*Junge 1963
1-5 d		*Jorgensen 1977a,b
2.3 d	Reaction with OH radical	*Cox 1975
3.5 d	Global sulphur budget	*Friend 1973

* In (NRCC, 1981)

APPENDIX IV.1: Meteorological input requirements

RCDM		SERTAD Prediction Module	OME	MESOPUFF II	RTM II/ADOM
Wind Speed	mean annual wind speed	joint frequency table describing wind speed and mixing height in each meteorological category	mean annual wind speed	gridded surface 850, 700, and 500 mb winds	gridded 3 dimensional, mass consistent wind field
Mixing Height	mean annual height	ratio of the frequency with which the wind is in the mean annual direction to the frequency assuming that all wind directions are equally likely	mean annual height		inversion height
Wind Direction	mean annual wind direction		mean annual wind direction		
Precipitation	mean length of wet and dry periods	rainfall categories at each model grid square and the mean dry period for each category	mean lengths of wet and dry periods	temporally and spatially varying precipitation	temporally and spatially varying, precipitation
Other Parameters				temperature, humidity and cloud cover	temperature, humidity, cloud cover

APPENDIX IV.2: Emissions requirements

RCDM		SERTAD Prediction Module	OME	MESOPUFF II	RTM II/ADOM
Species	SO ₂	SO ₂	SO ₂	SO ₂ , SO ₄ , NO _x	NO _x , SO ₂ , organics
Temporal Resolution	annual	seasonal or annual	annual		3 hours
Spatial Resolution	127 x 127 km grid square	127 x 127 km grid square at 4 different ranges of height:	latitude-longitude grid system	gridded emissions point and area sources at different levels	127 x 127 km gridded emissions point and area sources at different levels

APPENDIX IV.3: Physical and chemical parameterization data

RCDM	SERTAD Prediction Module	OME	MESOPUFF II	RTM II/ADOM
lifetime of primary and secondary pollutants under- going, - dry deposition - wet deposition lifetime of primary pollutants undergoing, - dry chemical changes - wet chemical changes	initial fraction of SO_2 converted to $\text{SO}_4^=$ rate of conversion of SO_2 to $\text{SO}_4^=$ scavenging coefficient of SO_2 and $\text{SO}_4^=$ SO_2 dry deposition velocity	SO_2 and $\text{SO}_4^=$ dry deposition velocity SO_2 wet scavenging rate $\text{SO}_4^=$ wet deposition velocity dry and wet conversion rate of SO_2 to $\text{SO}_4^=$ fraction of time of wet and dry deposition fraction of SO_2 and $\text{SO}_4^=$ to the total emissions released at the source	deposition velocity scavenging coeffi- cients chemical conversion constants gridded land use data (surface type and roughness)	scavenging coeffi- cients chemical conversion constants gridded land use data (surface type and roughness, topo- graphy)

APPENDIX IV.4: Diffusion parameters

RCDM		SERTAD Prediction Module	OME	MESOPUFF II	RTM II/ADOM
Horizontal	continental scale horizontal diffusivity	wind direction frequency (wind roses)	horizontal velocity fluctuations of synoptic scale turbulence	Gaussian puff expansion, dependent on stability.	parametrised according to Suragorinsky (1963).
Vertical	not used, instantaneous vertical mixing below mixing height	vertical eddy diffusivity	not used, instantaneous vertical mixing below mixing height	vertical eddy diffusivity, tabulated as function of stability	ADOM: analytical expressions dependent on heights and stability. RTMII: uses layer averaging, therefore treats only inter- layer fluxes.

APPENDIX V.1: Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg}\cdot\text{m}^{-3}$	Exposure	Injury	Yield
Alfalfa	130	5 $\text{hr}\cdot\text{d}^{-1}$ for 20 d	-	+ as much as 49%
	630-3140	3-5 $\text{hr}\cdot\text{wk}^{-1}$	-	no
	660	4 hr	no	no
	920	1 hr	-	no
	1970	3 hr	no	
	2100	2 hr	threshold dose for necrosis	-
	3930	3 hr	leaf necrosis	-
Apples	1050	6 hr	2%	
	1570	6 hr	+ 7.3% foliar + 5% leaf abscission	-
	6850	6 hr	+ 62% leaf abscission	+ 19% shoot growth
Barley	80-260	72 $\text{hr}\cdot\text{wk}^{-1}$ for growing season	-	no
	390	72 $\text{hr}\cdot\text{wk}^{-1}$ for growing season	-	+ 44%
	390-790	24 $\text{hr}\cdot\text{d}^{-1}$ x 7 d	severe	-

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	O_2		Effects on	
	Conc. $\text{mg}\cdot\text{m}^{-3}$	Exposure	Injury	Yield
Barley (cont'd)	590-790	2 hr	-	-
	630-3140	3-5 $\text{hr}\cdot\text{wk}^{-1}$	-	no
	790	5 $\text{hr}\cdot\text{d}^{-1}$ 6 $\text{d}\cdot\text{wk}^{-1}$ 12 wks	11%	+ 38%
	2620	2 hr	threshold for necrosis	-
Various Beans	92-465	8 hr	-	no
	390-790	24 $\text{hr}\cdot\text{d}^{-1}$ 7 d	severe foliar	-
	465	2 hr	slight	-
	520	continuous to maturation	-	+ 85%
	660	4 hr	no	no
	790	5 $\text{hr}\cdot\text{d}^{-1}$ 6 $\text{d}\cdot\text{wk}^{-1}$ 12 wks	<1%	+ 25%
	790	5 $\text{hr}\cdot\text{d}^{-1}$ 6 $\text{d}\cdot\text{wk}^{-1}$ 26 wks	2%	+ 15%
	2620	4 hr	25%	-
Corn	180-1390	24 $\text{hr}\cdot\text{d}^{-1}$	-	+ 7-24%
	390-790	24 $\text{hr}\cdot\text{d}^{-1}$ 7 d	severe	-

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg} \cdot \text{m}^{-3}$	Exposure	Injury	Yield
Corn (cont'd)	1310	100 hr	minimal foliar	no
Feed Grass	50-260	growing season	-	+ digest- ible dry matter after 2 years
	50-370	29 d and 22 d later	-	no
	63	85 d	-	+ 27% only at high nitrogen
	176	26 wks	88%	+ 52%
	178	103.5 $\text{hr} \cdot \text{wk}^{-1}$ for 20 wks	-	+ 28%
	290	4 wks	0-5%	+ 40-45%
	290	103.5 $\text{hr} \cdot \text{wk}^{-1}$ for 20 wks	-	+ 44%
	290	24 $\text{hr} \cdot \text{d}^{-1}$ 4 wks	-	+ 17-40% under high wind only
	320	9 wks	-	+ 46%
	340-2620	6 wks		0 - plants dead after 4 weeks
	380-1570	6 wks	-	+
	520	2 hr		-

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg} \cdot \text{m}^{-3}$	Exposure	Injury	Yield
Feed Grass (cont'd)	660	4 hr	no	no
	660-710	5-8 wks	-	+ 17-38%
	1310-28820	2 hr	-	-
	2620	4 hr	65% foliar	-
Oats	130-660	4 hr	no	no
	325-2620	1 and 3 hr	0-2%	-
Pea	260	18 d	-	+ 5%
	325-2620	1 and 3 hr	0-2%	-
	390	18 d	-	+ 8%
	660	18 d	no	+ 26%
Poinsettia	2620	3 hr	+	-
	5240	3 hr	+	-
	7860	1-3 hr	+	-
Radish	130-660	4 hr	no	-
	325-2620	1-3 hr	0-2%	-
	660	4 hr	no	no
	2620	4 hr	46%	-

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg} \cdot \text{m}^{-3}$	Exposure	Injury	Yield
Soybean	130-520	8 $\text{h} \cdot \text{d}^{-1}$ 9 $\text{d} \cdot \text{wk}^{-1}$ for 18 wks	-	no
	130-660	4 hr	no	no
	240-940	42 hr per 18 fumigations in 40 d	+	+ 6.4-15.9%
	260	6 hr/d for 43-133 d	no	+ 1% after 135 d
	310-2070	4.7/24 fumigations in 40 d	+	+ 12.3- 45.3%
	660	4 hr, 3 times per wk, for 11 wks	no	+ after 5 wks
	1310	1.5 hr	trace	no
	2100-5240	4.3 hr	-	+ 4.5-15%
	2620	1.5-3 hr	4% leaf injury	+ 5-29% short term only
	3930	3/4 - 3 hr	63-93% leaf injury	+ 24-94% short term only
Sunflower	180-1390	24 $\text{hr} \cdot \text{d}^{-1}$	-	+ 27-44%
	790	5 $\text{hr} \cdot \text{d}^{-1}$ 6 d/wk 12 wks	5%	+ 41%
	790	5 $\text{hr} \cdot \text{d}^{-1}$ 6 $\text{d} \cdot \text{wk}^{-1}$ 26 wks	16%	+ 29%

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg} \cdot \text{m}^{-3}$	Exposure	Injury	Yield
Tobacco	130	5 $\text{hr} \cdot \text{d}^{-1}$ for 20 d	-	+ 22% - no depends on ????
	130-660	4 hr	no	no
	180-1390	24 $\text{hr} \cdot \text{d}^{-1}$		+ 44%
	660	4 hr	1% leaf injury	no
Tomato	520	15 d	threshold dose for initial symptoms	-
	660	4 hr	no	no
	2620	4 hr	33%	-
Wheat	80-260	72 hr per wk for growing season	-	no
	100	3 hr x 8 exp.	no	no
	390	72 hr/wk for growing season	-	+ 42% Durum W. no Spring W.
	520	30, 78, 100 hr	little	no
	520-790	2 hr	little	no
	630-3140	3 hr per wk to 3 hr per 5 wks	-	no

APPENDIX V.1 (cont'd): Effects of SO_2 on various agronomic and horticultural crops
(Based on EPA 1982a)

Plant	SO_2		Effects on	
	Conc. $\text{mg} \cdot \text{m}^{-3}$	Exposure	Injury	Yield
Wheat (cont'd)	1050	30, 78, 100 hr	little	no
	1180	3 hr x 7 exp. for growing season	-	no
	1570	30, 78, 100 hr	little	no

no = no effects on yield; no effect on rate of injury.

- = no data.

† = yield reduced (eg. † up to 49%; yield is reduced by as much as 49%).

‡ = rate of injury increased; yield increased.

APPENDIX V.2: Effect of Acid Deposition on Various Crops
(Based on EPA, 1984; Data of Field Research)

Plant	Deposition $\text{kg} \cdot \text{ha}^{-1}$		pH	Exposure	Effect on Yield
	$\text{SO}_4^=$	NO_3^-			
Alfalfa	2.13-120.84	2.26	5.6-3.0	1.5 hr per event x 26 events	+ 9% pH 4 compared to pH 5.6
	2.13- 89.15	2.26- 60.85	5.6-3.0	1.5 hr per event x 26 events	no
Beet	0.08- 7.09	0.2	5.7-2.7	0.001 hr per event x 19 events	+ all pH's compared to pH 5.6
Corn	4.03-198.16	4.75-135.47	5.6-3.0	1.5 hr per event x 58 events	+ 9% pH4 compared to pH 5.6
Fescue	2.13-120.84	2.26	5.6-3.0	1.5 hr per event x 26 events	+ 19-24% compared to pH 5.6
	2.13- 89.15	2.26- 60.85	5.6-3.0	1.5 hr per event x 26 events	no
Bean	13.02-98.05	5.57	6.0-3.2	0.17 hr per event x 27 events	no
Mustard	2.76-56.33	1.95-38.04	5.6-3.0	1.5 hr per event x 16 events	+ 31 - 33% compared to pH 5.6
Radish	0.43-33.4	0.19-16.6	4.2-2.8	1 hr per event x 6 events	+ 3-17% compared to pH 5.6
	0.36-27.8	0.16-13.8	2.8-5.6	1 hr per event x 5 events	+ 7-13% compared to pH 5.6

APPENDIX V.2: (con'd): Effect of acid deposition on various crops
(Based on EPA, 1984; Data of Field Research)

Plant	Deposition ($\text{kg} \cdot \text{ha}^{-1}$)		pH	Exposure	Effect on Yield
	$\text{SO}_4^=$	NO_3^-			
Radish (cont'd)	0.64-41.0	0.91-28.1	5.6-3.0	1.5 hr per event x 12 events	+ 25% pH 3.5 compared to pH 5.6
	0.04-3.4	0.1	5.7-2.7	0.01 hr per event x 9 events	no
Soybean	0.2-97.2	0.56-59.6	2.3-4.1	0.001 hr per event x 41-29 events	+11.5% pH 2.7 compared to pH 4.1
	2.88-39.97	0.32-4.49	4.0-2.8	1 hr per event 13-18 events	+ 32% pH 2.8 compared to pH 4
	6.19-683.91	3.06-63.31	5.4-2.4	0.5 hr per event x 26 events	no
	8.9-18.72	2.82-4.26	3.06-5.6	0.33 hr per event x 11-15 events	no
	2.88-39.97	0.32-4.49	4.0-2.8	1 hr per event x 18 events	+ 24% pH 2.8 compared to pH 4
Spinach	2.72-71.18	1.91-35.71	5.6-3.0	1.5 hr per event x 15 events	no

+ = yield increased.

+ = yield decreased.

**APPENDIX V.3: Effect of acid deposition on various crops
(Based on EPA, 1984)**

Plant	Deposition ($\text{g}\cdot\text{m}^{-2}$)		pH	Exposure	Effects on Yields
	$\text{SO}_4^{=}$	NO_3^{-}			
Alfalfa	0.3-54.92	0.417	5.6-3.0	1.5 hr per event x 56 events	+ 31% only at pH 3.5
Barley	0.238-44.13	0.335	5.6-3.0	1.5 hr per event x 45 events	no
Beet	0.14-25.5	0.193	5.6-3.0	1.5 hr per event x 26 events	+ 43% only at pH 3.0
Bluegrass	0.382-70.61	0.472	5.6-3.0	1.5 hr per event x 72 events	no
Broccoli	0.117-21.58	0.164	5.6-3.0	1.5 hr per event x 22 events	+ 25% only at pH 3.0
Bean	0.001-0.041	0.001	5.6-3.2	0.67 hr per event x 16-18 events	no
Cabbage	0.27-50.02	0.379	5.6-3.0	1.5 hr per event x 51 events	no
Cauliflower	0.122-22.56	0.171	5.6-3.0	1.5 hr per event x 23 events	no
Corn	0.11-19.61	0.149	5.6-3.0	1.5 hr per event x 20 events	+ 13% only at pH 3.0
Fescue	0.31-57.86	0.439	5.6-3.0	1.5 hr per event x 59 events	no

APPENDIX V.3 (cont'd): Effect of acid deposition on various crops
(Based on EPA, 1984)

Plant	Deposition ($\text{g}\cdot\text{m}^{-2}$)		pH	Exposure	Effects on Yields
	SO_4^-	NO_3^-			
Pea	0.15-27.46	0.208	5.6-3.0	1.5 hr per event x 28 events	no
Pepper	0.20-37.27	0.283	5.6-3.0	1.5 hr per event x 38 events	† 20% only at pH 3.0
Bean	0.007-18.14	0.043	6.0-3.2	0.17 ha per event x 24 events	† 75% when treatment intervals were pH 6.0/3.2/6.0 and pH 6.0/3.2/3.2 † 50% at pH 3.2 ‡ 50%, when treatment intervals were pH 3.2/6.0/6.0 and pH 6.0/6.0/3.2
Lettuce	0.023-1.383	0.029-1.786	5.7-3.2	2 hr per event x 3 events	† 7-10% at pH 3.2 for $\text{SO}_4^-/\text{NO}_3^- > 0.82$
Mustard Green	0.074-13.73	0.104	5.6-3.0	1.5 hr per event x 14 events	† 14% at pH 4.0 No effect at pH 3.5 † 31% at pH 3.0

APPENDIX V.3 (cont'd): Effect of acid deposition on various crops
(Based on EPA, 1984);

Plant	Deposition ($\text{g}\cdot\text{m}^{-2}$)		pH	Exposure	Effects on Yields
	$\text{SO}_4^{=}$	NO_3^{-}			
Oats	0.254-47.07	0.357	5.6-3.0	1.5 hr per event x 48 events	no
Onion	0.34-63.75	0.484	5.6-3.0	1.5 hr per event x 65 events	no
Orchard Grass	0.19-34.32	0.26	5.6-3.0	1.5 hr per event x 35 events	+ 23% only at pH 3.0
Pinto Bean	8.533-258.16	1.365	5.7-2.5	0.33 hr per event x 45 events	+ 28% at pH 2.9 + 29% at pH 2.7 + 39% at pH 2.5
Potato	0.276-51.00	0.387	5.6-3.0	1.5 hr per event x 52 events	+ 11% at pH 3.5 + 8% at pH 3.0
Radish	0.064-11.77	0.089	5.6-3.0	1.5 hr per event x 12 events	+ at pH 3.5-3.0
Red Clover	0.300-54.92	0.417	5.6-3.0	1.5 hr per event x 56 events	no
Raje Grass	0.31-56.88	0.432	5.6-3.0	1.5 hr per event x 58 events	no

APPENDIX V.3 (cont'd): Effect of acid deposition on various crops
(Based on EPA, 1984)

Plant	Deposition ($\text{g}\cdot\text{m}^{-2}$)		pH	Exposure	Effects on Yields
	$\text{SO}_4^{=}$	NO_3^{-}			
Spinach	0.074-13.73	0.104	5.6-3.0	1.5 hr per event x 14 events	no
Soybean (Amsoy 71)	9.755-295.12	1.561	5.7-2.5	0.17 hr per event x 78 events	+ 11% at pH 3.1 + 11% at pH 2.5
Soybean (Wells)	0.669-29.40	0.721-9.80	5.6-3.0	0.33 hr per event x 10 events	No effect except + 25% at pH 3.0
Strawberry	0.42-78.46	0.595	5.6-3.0	1.5 hr per event x 80 events	+ 51% at pH 4.0 + 72% at pH 3.5 or 3.0
Tobacco	0.127-23.537	0.179	5.6-3.0	1.5 hr per event x 24 events	no
Tomato	0.27-50.02	0.379	5.6-3.0	1.5 hr per event x 51 events	+ 31% only at pH 3.0
Wheat	0.244-45.11	0.342	5.6-3.0	1.5 hr per event x	no
Red Clover	0.300-54.92	0.417	5.6-3.0	46 events	

† = yield increased.

‡ = yield decreased.

MAPS NOT

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